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CONTENTS

ACCURACY OF ESTIMATING THE SPECIES AND SIZES OF OSPREY PREY: A TEST OF METHODS. David N. Carss and J.D. Godfrey.....	57
LONG-TERM POPULATION MONITORING OF OSPREY ALONG THE UMPQUA RIVER IN WESTERN OREGON. Joseph W. Witt	62
POSSIBLE SECOND CLUTCHES IN A MEDITERRANEAN MONTANE POPULATION OF THE EURASIAN KESTREL (<i>FALCO TINNUNCULUS</i>). Juan A. Fargallo, Guillermo Blanco and Eduardo Soto-Largo.....	70
MEXICAN SPOTTED OWL HABITAT CHARACTERISTICS IN ZION NATIONAL PARK. Sarah E. Rinkevich and R.J. Gutiérrez.....	74
FLEDGING AND MIGRATION OF JUVENILE BALD EAGLES FROM GLACIER NATIONAL PARK, MONTANA. B. Riley McClelland, Patricia T. McClelland, Richard E. Yates, Elaine L. Caton and Mary E. McFadzen.....	79
INTRA-YEAR REUSE OF GREAT HORNED OWL NEST SITES BY BARN OWLS IN EAST-CENTRAL COLORADO. David E. Andersen.....	90
A COMPARISON OF BEHAVIOR AND SUCCESS RATES OF MERLINS AND PEREGRINE FALCONS WHEN HUNTING DUNLINS IN TWO COASTAL HABITATS. Joseph B. Buchanan.....	93
SHORT COMMUNICATIONS	
HABITAT PREFERENCE OF CRESTED SERPENT EAGLES IN SOUTHERN JAPAN. Mutsuyuki Ueta and Jason S. Minton.....	99
A POSSIBLE CASE OF POLYANDRY IN MONTAGU'S HARRIER. Beatriz Arroyo	100
NOTES ON THE DIET OF SHORT-EARED OWLS (<i>ASIO FLAMMEUS</i>) IN TEXAS. Kelly M. Hogan, Morgan L. Hogan, Jennifer Gable and Martin Bray.....	102
LETTERS	105
BOOK REVIEWS. Edited by Jeffrey S. Marks	108
THESIS ABSTRACT	110

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ACCURACY OF ESTIMATING THE SPECIES AND SIZES OF OSPREY PREY: A TEST OF METHODS

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ABSTRACT.—The accuracies of examining uneaten prey remains collected at feeding sites and of directly observing fish captured while birds forage, common methods of determining the species composition and size structure of prey in the diets of ospreys (*Pandion haliaetus*), were tested during the summer of 1992 at two shallow lakes in northeastern Scotland. Prey remains were collected below feeding perches and the number of heads and paired jaws was used to estimate the minimal number of each species in the diet. Key cranial bones were used for species identification and length estimation. Direct field observations were also made to identify the species and sizes of fish taken by foraging ospreys. Fish species were identified by body shape and lengths were estimated by comparison with the size of the ospreys. The accuracy of field observations was tested experimentally using a life-sized model osprey and a selection of northern pike (*Esox lucius*) and perch (*Perca fluviatilis*) of various sizes. Results showed that the analysis of prey remains gave an accurate estimation of the size range of osprey prey, although small fish (<25 cm) were underrepresented. Tests of field observations showed that most fish could be correctly identified on the basis of their body shape but there were consistent inter-observer differences in fish length estimations. These differences should be considered in studies using field estimates of prey size, particularly those involving energetic calculations where small errors in length estimations can lead to large errors in estimations of mass and, hence, energy.

KEY WORDS: *osprey*; *Pandion haliaetus*; *diet*; *prey estimation*; *field techniques*.

Exactitud de la estimación de tipos y tamaños de presas de *Pandion haliaetus*: una prueba de métodos

RESUMEN.—Las exactitudes de examinar presas no comidas que permanecen en los comederos y de observaciones directas de peces capturados mientras las aves se alimentan, métodos comunes de determinación de la composición de especies y tamaño de la estructura de presa en la dieta de *Pandion haliaetus*, fueron probados durante el verano de 1992 en dos lagos superficiales al noreste de Escocia. Los restos de las presas fueron colectados bajo comederos; el número de cráneos y pares mandibulares fueron usados para estimar el número mínimo de cada especie en la dieta. Claves de huesos craneales se usaron para la identificación de especies y estimación de longitud. También se hicieron observaciones de terreno para identificar los tipos y tamaños de peces capturados por águilas pescadoras. Las especies de peces fueron identificadas por la forma del cuerpo y la longitud fue estimada por comparación con el tamaño de la misma águila. La exactitud de las observaciones de campo fue probada experimentalmente usando un modelo "life-sized" del águila pescadora y una selección de varios tamaños de *Esox lucius* y *Perca fluviatilis*. Los resultados mostraron que el análisis de restos de presa entregan una estimación exacta del rango de tamaño de las presas del águila, aunque los peces pequeños fueron subrepresentados. Pruebas de observaciones de campo, mostraron que la mayoría de los peces podría ser identificado correctamente sobre la base de su forma corporal, en cambio hubo consistentes diferencias

entre observadores respecto a las estimaciones del largo. Estas diferencias podrían ser consideradas en estudios usando estimaciones de campo del tamaño de presa, particularmente aquellas que envuelven cálculos energéticos donde pequeños errores en las estimaciones de longitud podrían llevar a cometer grandes errores en estimaciones de masa y por lo tanto de energía.

[Traducción de Ivan Lazo]

The species composition and size structure of osprey (*Pandion haliaetus*) prey have been determined by collecting uneaten prey remains at nests and feeding perches, and by directly observing fish taken while ospreys forage (Poole 1989). There are potential biases associated with each method. Using the first, the frequency of small fish may be underestimated in the diet if, for example, they are completely ingested or their remains are hard to find. Conversely, overestimates may occur if large items are removed preferentially by scavengers such as corvids or foxes. The second method may also be biased because field identification and size-estimation of fish may be inaccurate (see discussion in Carss and Brockie 1994 for osprey and also Bayer 1985, Cezilly and Wallace 1988 for other species). In this study, we tested the errors associated with both methods of assessing osprey diets.

STUDY AREA AND METHODS

Data on osprey prey were collected at two lakes, Loch Davan (42 ha) and Loch Kinord (82 ha), in the Dinnet National Nature Reserve in northeast Scotland from June–August 1992. Pelagic fish species in these shallow (mean depth = 1.2 and 1.5 m, respectively), “kettle-hole” lochs were principally northern pike (*Esox lucius*) and perch (*Perca fluviatilis*). The only other fish was the common eel (*Anguilla anguilla*). The northern pike is a common top predator of freshwater ecosystems in Europe and North America and often found in association with perch; such simple fish communities are relatively common in Scotland.

Prey remains were collected below feeding sites (mainly telegraph poles but also trees) throughout the reserve and in adjacent areas. The number of heads or paired jaws was taken as the minimal number of each species in the diet and key cranial bones were extracted for species identification and length estimation following Carss and Brockie (1994).

Direct field observations of foraging ospreys were made by one observer (JDG) from June–August 1992 and all daylight hours from 0515–2230 H were sampled in a variety of weather conditions. Individual, foraging ospreys were watched from the loch shore with 8×32 bin-

oculars or a 15–65×70 telescope. Dives were classed as successful if a fish was seen to be carried away and unsuccessful if no fish was carried. The species of fish taken was identified from its body shape and its length was estimated by comparison with the size of the ospreys.

The accuracy of direct field observations of osprey prey was tested at the Institute of Terrestrial Ecology, Banchory, using a life-size model osprey (body = 55 cm, wing span = 155 cm) and a selection of pike and perch of various sizes that were caught in the study lochs. Fish were suspended between the talons of the model osprey which was then raised approximately 5 m into the air for a period of 10–20 sec. The model was observed against the sky from a distance similar to that encountered in the field (ca. 150 m). Ten pike (fork lengths [FL] = 10, 11, 12, 21, 21, 36, 36, 39, 40, 50 cm) and three perch (FL = 8, 9, 12 cm) were shown, 10 of which were presented twice. Fish were presented in arbitrary order and observers had no prior knowledge of the range of sizes to be expected. At some point during the trial, the model osprey was shown without a fish, giving a total of 24 presentations. Six observers, including the two authors, took part in the tests for a total of 144 observations. Data were analyzed by linear regression of the relative errors in the estimated fork lengths ($[\text{estimated} - \text{actual}]/\text{actual}$) on the actual fork lengths of fish presented to each observer. We tested for differences in either the slopes or the intercepts of each observer's estimation equation assuming (a) a different slope and a common intercept or, (b) a different intercept and a common slope for each observer.

RESULTS AND DISCUSSION

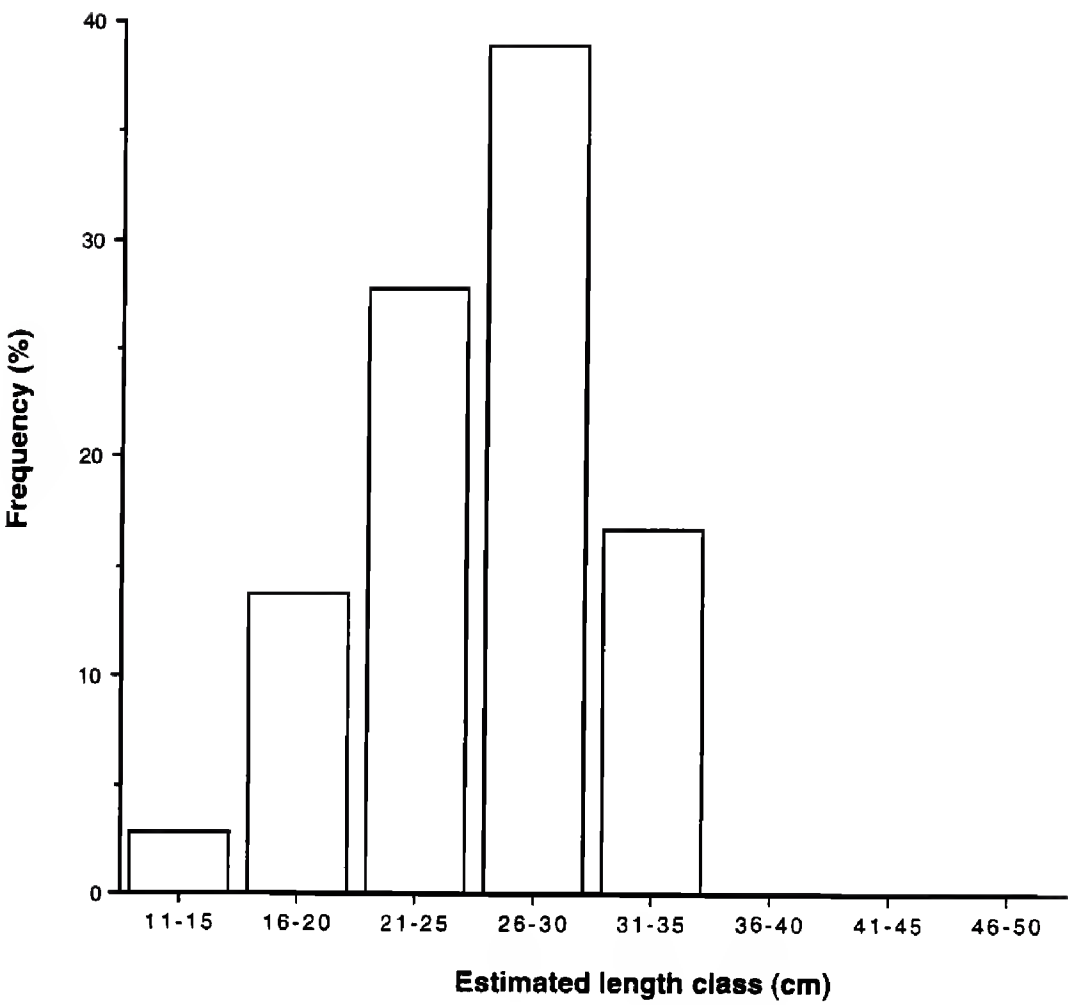
Remains of 101 individual fish were collected between June–August. The majority of remains collected were fish heads, although some tails and intact carcasses were also found. Remains were mostly those of pike (64%) with the remainder being perch, as was expected given the simple fish community of the lakes. In general, piscivorous fishes are seldom found in the diets of ospreys (reviewed in Poole 1989). Perch and pike comprise no more than 16% and 37%, respectively, of the diet of European ospreys (Cramp and Simmons 1980).

It was clear that ospreys took a particular size-

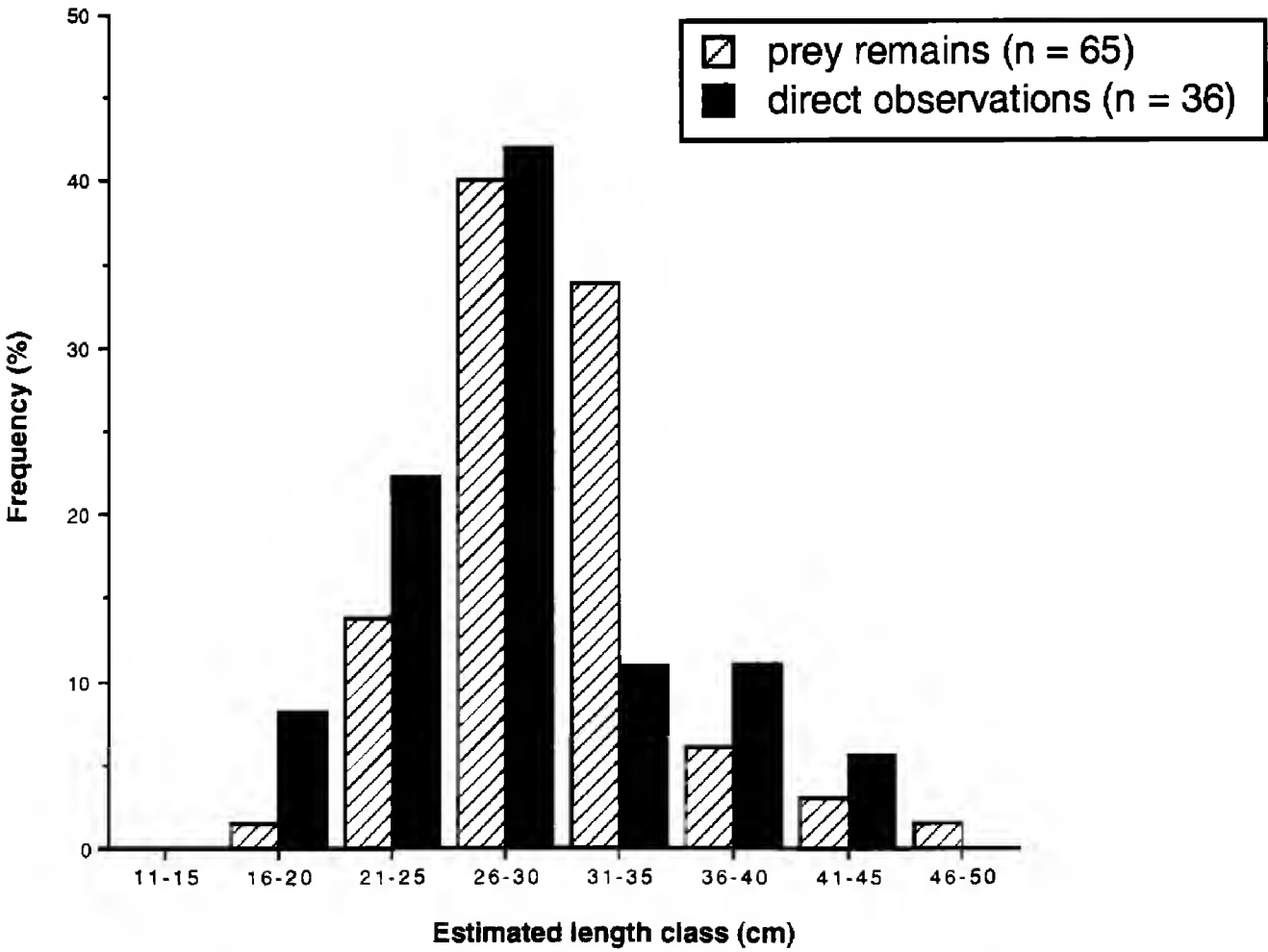
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Figure 1. (a) Length classes of perch ($N = 36$) estimated from prey remains collected from feeding sites. (b) Length classes of pike estimated from both prey remains ($N = 65$) and field observations ($N = 36$) of foraging ospreys. Data collected from Dinnet National Nature Reserve, June–August 1992.

(a) Perch (n = 36)



(b) Pike



range of prey at Dinnet. Length estimates for pike ranged from 19–46 cm and those for perch were from 12–36 cm (Fig. 1a). These size ranges were similar to those reported by Cramp and Simmons (1980) and Poole (1989), and strikingly similar to those estimated using the same method in central Scotland (perch: range = 18–30 cm, $N = 16$; pike: range = 24–44 cm, $N = 25$) (Carss and Brockie 1994).

We observed 38 fish actually captured by foraging ospreys. All but two, perch with estimated lengths of 18 and 25 cm, were pike. Length estimates for pike (Fig. 1b) ranged from 16–44 cm ($\bar{x} = 27$ cm, $SE = 11$, $N = 36$). Overall, size ranges determined using this method were similar to those obtained using prey remains with the largest proportion of fish taken in the 26–30 cm range. Although not statistically significant (χ^2 test on numbers of fish remains and observations in <25 cm, 26–35 cm, and >36 cm size classes), small pike (<25 cm) were less frequently observed in prey remains than during direct field observations at the lochs, and fewer large pike (>30 cm) were seen taken than were represented in remains collected at nearby feeding perches. We concluded that estimates of osprey diets from prey remains probably gave a biased picture of the lengths of fish taken with the proportions of small fish being underrepresented.

We have found the undigested remains of fish up to 12 cm long in the guts of larger piscivorous fish that were partially eaten by ospreys. Therefore, the presence of small fish remains at nests or feeding sites does not necessarily imply that fish of this size have been taken directly by ospreys; such a phenomenon could explain the record of a 4 cm fish at the nest (McLean and Byrd 1991).

During field tests with the model osprey, all six observers were able to correctly determine when the osprey was not carrying a fish. Most fish (92.8%) in the remaining 138 experimental trials were correctly identified to species (5.8% misidentified and 1.5% unidentified). The eight misidentified fish (4 pike and 4 perch) were the smallest fish used in the trials (\bar{x} FL = 14 cm, $SE = 2.4$, range = 8–21 cm). Presumably, larger fish were correctly identified more often because of differences in their body shape, with pike tending to be elongate and perch deep-bodied. Observations of actual prey captures by ospreys usually last longer than 20 sec and real ospreys carry live fish which hold their fins erect increasing the opportunity for

Table 1. Percentages of osprey prey length estimates correctly and incorrectly assigned to arbitrary 5 cm size categories by each of six observers (a–f). Observers differed in their ability to correctly categorize estimations ($\chi^2 = 11.03$, $df = 5$, $P = 0.05$).

LENGTH ESTIMATE	OBSERVER					
	a	b	c	d	e	f
% Correct	34.8	52.2	43.5	21.7	26.1	60.9
% Incorrect	65.2	47.8	56.5	78.3	73.9	39.1
Total estimates	23	23	23	23	23	23

prey identification. Nevertheless, the accuracy of identifications may be reduced in other areas where confusion could arise between similarly-shaped fishes such as perch and roach (*Rutilus rutilus*), or pike and salmonids (*Salmo* spp., *Oncorhynchus* spp.).

The regression analysis showed that there was significant variation among observers in the estimation of fish sizes. Both the intercepts ($F_{5,126} = 10.7$, $df = 5$, $P < 0.001$) and slopes ($F_{5,126} = 13.4$, $df = 5$, $P < 0.001$) of observer regression lines differed significantly. We therefore concluded that such differences should be taken into account in studies relying on length estimates in the field.

Most (71%) of the 138 estimates were within 20% of the true lengths with those of one observer (JDG) being consistently within 10% of the actual lengths. Most observers estimated fish lengths within 3–9 cm of the actual length and one observer (JDG) estimated them with 2–4 cm accuracy. These values would likely be the same under actual field conditions for a similar observation distance. After length estimates were assigned to arbitrary 5 cm size classes (e.g., 6–10 cm, 11–15 cm), we were unable to improve observer accuracy and 39–78% of the estimates were still incorrectly assigned (Table 1). A further increase in the range of size classes used would increase the proportions of estimated lengths correctly identified, but such results would be increasingly less meaningful. Therefore, it is recommended that observers be tested before making size estimations of osprey prey in the field.

The experimental trials suggested that field observations of fish taken by foraging ospreys would give an accurate estimate of the proportions of each prey species in the diet but that size estimates of fish would be less reliable because some observers were able to estimate the lengths of fish more

accurately than others. This may have important implications for energetic studies where prey mass, rather than its length, is a crucial factor. Because body mass varies as the cube of length, small errors in length estimation will lead to large errors in the estimation of mass.

We found that accurate length estimates could be obtained from the collection of prey remains at feeding sites. While this also appeared to be a valid technique for estimating the size range of osprey prey, it underestimated the proportion of small fish (<25 cm) taken. Nevertheless, this method was far less labor intensive and, hence, cheaper, than direct observations in determining the diets of ospreys.

ACKNOWLEDGMENTS

We would like to thank Mike Harris, Hans Kruuk, Mick Marquiss and Sarah Wanless for acting as observers during the direct observation trials, and Hans and Phil Bacon for collecting some of the prey remains in the field. Mick, Sarah and Ken Nelson commented on an earlier

draft of the manuscript, as did Alan Poole and Peter McLean. Dave Elston provided statistical advice on the analysis of results from the experimental trials while Jim Parkin gave us additional osprey observations and provided on-site accommodation for one of us (JDG).

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LONG-TERM POPULATION MONITORING OF OSPREY ALONG THE UMPQUA RIVER IN WESTERN OREGON

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ABSTRACT.—From 1981–90, the osprey population along the Umpqua River between Roseburg and Reedsport, Oregon increased by 153% (17% annual rate). The first observed decrease in the population occurred in 1991 when one previously occupied breeding territory became vacant. Management activities on USDI Bureau of Land Management administered lands within the study area between 1981–88 consisted of the installation of 24 nesting platforms and 17 accessory perches. During this study, 15 of the platforms were occupied by breeding ospreys accounting for over 40% of the total population increase along the Umpqua River. Productivity surveys using either ground survey (1981 and 1984) or helicopter survey (1982, 1983, 1985–91) techniques estimated an average productivity of 1.21 (range = 0.87–1.86) fledglings/occupied territory, 1.33 (range = 0.93–1.93) fledglings/breeding attempt, and 2.04 (range = 1.50–2.47) fledglings/successful breeding attempt. Platform sites were more productive than natural nest substrates but the difference was not significant. The observed rate of increase of the osprey population between 1981–1990 was similar to that reported elsewhere where nest platforms have been installed to increase osprey numbers.

KEY WORDS: *osprey*; *Pandion haliaetus*; *population monitoring*; *reproduction*; *artificial platforms*; *Oregon*.

Monitoreo poblacional a largo plazo de *Pandion haliaetus* a lo largo del Río Umpqua al Oeste de Oregon

RESUMEN.—Desde 1981 a 1991, las poblaciones de *Pandion haliaetus* a lo largo del Río Umpqua entre Roseburg y Reedsport, Oregon, han aumentado en un 153% (tasa anual: 17%). La primera disminución poblacional ocurrió en 1991, cuando un territorio reproductivo ocupado previamente quedó vacío. Las actividades de manejo del “USDI Bureau of Land Management,” que administró tierras en el área de estudio durante 1981 a 1988, consistieron en la instalación de 24 plataformas de nidificación y 17 perchas accesorias. Durante este estudio, 15 de las plataformas fueron ocupadas por *P. haliaetus* reproductivos, aumentando sobre el 40% la población total a lo largo del Río Umpqua. Con técnicas de estimación de productividad vía rutas terrestres (1981 y 1984) o aéreas (1982, 1983, 1985 a 1991), se estimó una productividad promedio de 1.21 (rango = 0.87–1.86) volantones/territorio ocupado, 1.33 (rango = 0.93–1.93) volantones/esfuerzo reproductivo y 2.04 (rango = 1.50–2.47) volantones/esfuerzo reproductivo exitoso. Las plataformas fueron más productivas que los sustratos de nidificación naturales, pero la diferencia no fue significativa. La tasa de incremento poblacional observada para el águila pescadora entre 1981 y 1990, fue similar a otros sitios en los que se han instalado plataformas para incrementar el número de águilas pescadoras.

[Traducción de Ivan Lazo]

Population status and productivity of ospreys (*Pandion haliaetus*) was an issue of concern and study during the late 1960s and early 1970s as evidence mounted indicating that pesticides, specifically DDT, were influencing the survival and reproductive success of some fish-eating raptors (Ames 1966, Ames and Mersereau 1964, Keith 1966, Henny and Wight 1969, Ratcliffe 1967, Anderson and Hickey 1972, Henny 1977, Reese 1977, Spitzer et al. 1977) and peregrine falcons (*Falco peregrinus*) (Ratcliffe 1969, Snow 1972). More recently, the fo-

cus of studies has shifted toward understanding the general ecology of ospreys (Swenson 1978, Jamieson and Seymour 1983, Poole 1989) and effects of weather (Stinson et al. 1987, Machmer and Ydenberg 1990), foraging, and courtship feeding behavior on their reproduction (e.g., Poole 1985, Hagan and Walters 1990).

In 1981, the U.S. Department of the Interior, Bureau of Land Management (BLM) initiated a nesting platform project along the Umpqua River with the intent of enhancing nesting habitat for ospreys

(Witt 1990), thereby facilitating the recruitment of new breeders into the population (Postupalsky 1978), and potentially mitigating some of the historic impacts from nest tree loss. This study was conducted between 1980–91, to assess the utility of the nesting platforms, examine the patterns of osprey productivity, and assess population trends during the study period along the Umpqua River, Oregon.

METHODS

The study area along the Umpqua Rivers consisted of a 3 km wide and 154 km long transect between Roseburg and Reedsport, Oregon. Approximately 24 km of the area surveyed was along the North Umpqua and South Umpqua Rivers (see Fig. 1, Witt 1990). The dominant nesting habitat or nesting substrate along the Umpqua River occurred in the Western Hemlock Zone (*Tsuga heterophylla*), with a smaller portion of the study area being in an Oregon White Oak (*Quercus garryana*) community (Franklin and Dyrness 1973).

Based on the observed distribution of both occupied and unoccupied sites in 1980–81, 24 platforms and 17 accessory perch trees were installed between 1981–89. Installation of nesting platforms was based on availability of BLM administered lands within 400 m of the Umpqua River, the distance from occupied nests, proximity to foraging areas, and availability of live trees with a diameter at breast height (dbh) of at least 125 cm. Perch trees were created when it was subjectively determined that there was an inadequate number of perch sites available at the platform site. In a few instances, trees near the platform or perch trees were pruned to increase visibility from the site to adjacent water.

Trees selected for the placement of platforms were topped 5–8 cm above a whorl of limbs where the diameter of the tree was 12–15 cm (usually 46 m above the ground). All lateral limbs were pruned for 8–12 m below each platform and were cut 0.9–1.2 m from the bole of the tree; thereby, creating a visual appearance of a snag. Perch trees were treated in similar manner except that perch trees were topped where the diameter of the tree was between 6–9 cm.

Platforms were constructed using four 1.22 m western red cedar (*Thuja plicata*) 2 × 4's in a crisscross pattern producing an internal 0.6 × 0.6 m cup; two shorter 2 × 4's (0.92 m long) were placed in the center to anchor the platform to the top of the tree (see Poole 1989, Fig. 10.1 and Witt 1990, Fig. 2). Also, between the shorter 2 × 4's and the longer 2 × 4's of the platform, a 16½ gauge 2" × 2" wire was sandwiched in and secured to prevent egg loss through the nest (Ames and Mersereau 1964).

From 1980–91, the osprey nesting population within the study area was monitored by surveying the study area twice a year. The first visit, usually a ground survey from the existing road system for historically and newly occupied sites occurred during the first or second week in May. Categorization of each site was based on terminology used by Postupalsky (1974). An occupied territory was a site with nest and a pair of ospreys present, a breeding attempt was one in which eggs were present in the

nest or where an adult bird was seen in incubating position, and a successful breeding attempt was a site where at least one young was raised to legal banding age. Productivity estimates were based on the number of young raised to banding age, and was calculated for occupied territories, breeding attempts, and successful breeding attempts. The second visit was a productivity survey and was completed during the last week of June in 1982–83 and 1985–91 using a helicopter. Due to fiscal constraints in 1981 and 1984, the surveys were conducted from the ground using a spotting scope during the first two weeks of July. If, during the helicopter survey, a nest contained birds that were not close to fledging, a third visit was made prior to fledging.

To determine the influence platforms may have had on productivity during the study, all breeding attempts were classified as either on natural or artificial substrates and their productivity was pooled for all years and analyzed using a one-tailed t-test and a one-tailed variance ratio test (Zar 1974). To examine population trends during the study the annual percent increase in the population was analyzed using log-linear regression.

RESULTS

The availability and use of platforms increased gradually during the study period, with 12.5% and 52.5% of the platforms occupied by breeding ospreys in 1981 and 1990, respectively. During the latter part of the study, ospreys began using standard wooden power poles ($N = 2$) and modified power poles ($N = 3$) erected by a local power company. In 1989 and 1990, breeding attempts on artificial structures (both platforms and power poles) represented 33.3% and 38.6% of the total breeding population in the study area, respectively.

Bald eagle (*Haliaeetus leucocephalus*) breeding territories between Roseburg and Elkton increased from two to five occupied sites during the study period. Two of the three new sites were on osprey platforms. In each case, eagles used platforms occupied by successfully breeding ospreys the year previous. After nesting 1 yr on these platforms, the eagles moved into adjacent forest stands and established nests in the lower crown of live trees. During the study period, only one bald eagle breeding attempt on platforms was successful fledging one eagle.

The number of occupied territories increased from 17 in 1981 to a high of 43 territories in 1990 for a 153% increase in the osprey population along the river (Fig. 1a). In 1991, the first decrease in the osprey population occurred when the number of occupied sites decreased by one site (Fig. 1b). Log-linear regression analysis of the increase indicated that there was a significant increase in the population ($R^2 = 0.804$, $F = 32.89$, $P < 0.0005$).

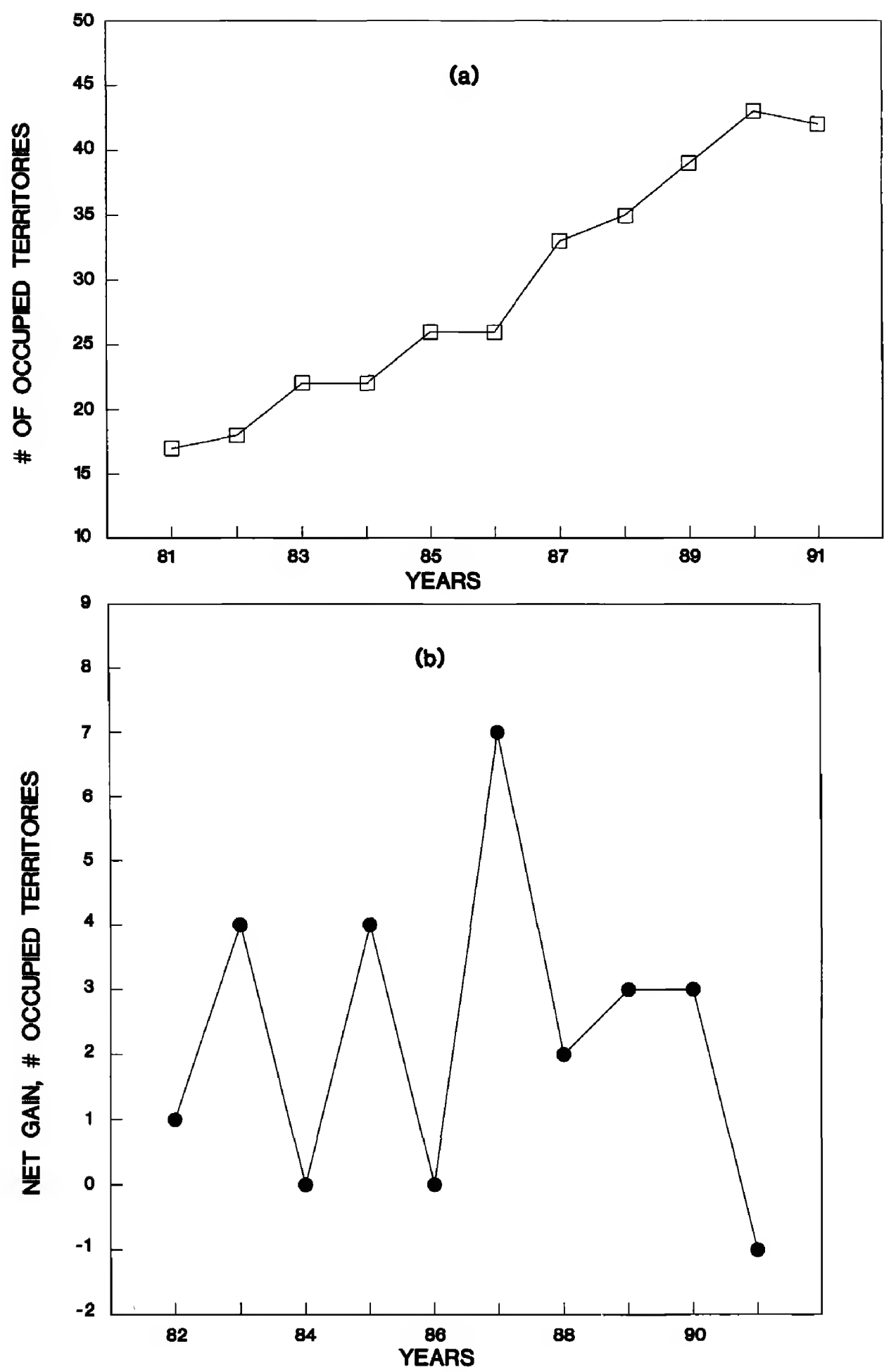


Figure 1. Osprey population changes within the Umpqua River study area. (a) The observed number of occupied territories between 1981–91. (b) The observed number of net gains or losses in the number of occupied territories between 1982–91.

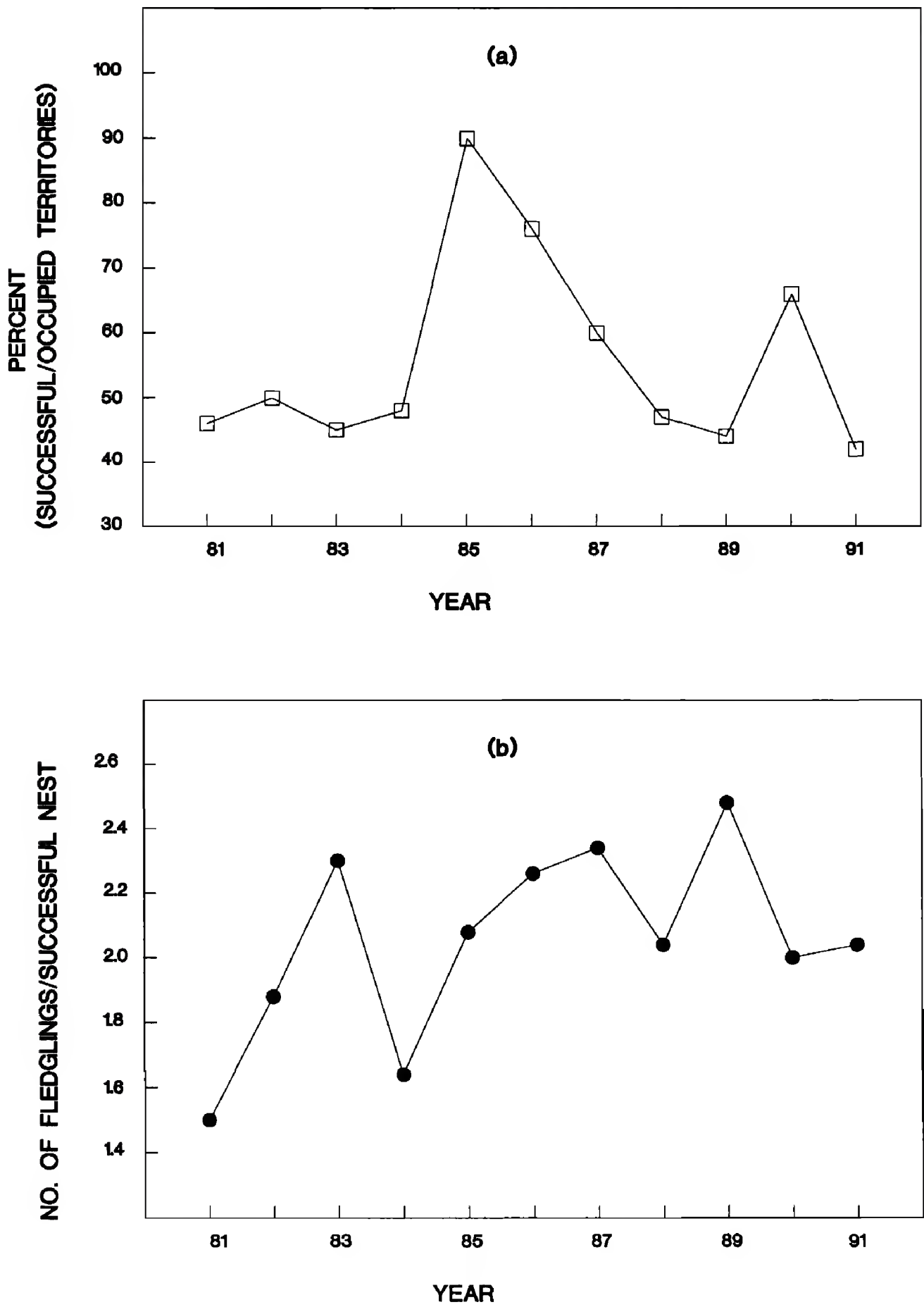


Figure 2. Breeding success of the ospreys along the Umpqua River. (a) The percent of the occupied territories that were successful between 1981–91. (b) The mean number of fledglings produced per successful breeding attempt.

Similarly, the number of breeding attempts increased from 14 in 1981 to 41 in 1990. This increase was also irregular and the overall pattern changed in 1991 when the number of breeding attempts decreased by two.

Overall, the mean productivity of the osprey

population during the study was 1.21 fledglings per occupied territory (range = 0.87–1.86, $N = 303$) and 1.33 fledglings per breeding attempt (range = 0.93–1.93, $N = 193$, Fig. 2). Only 17 fledglings were produced in 1982 but the number increased to a high of 62 fledglings in 1990. The percent of oc-

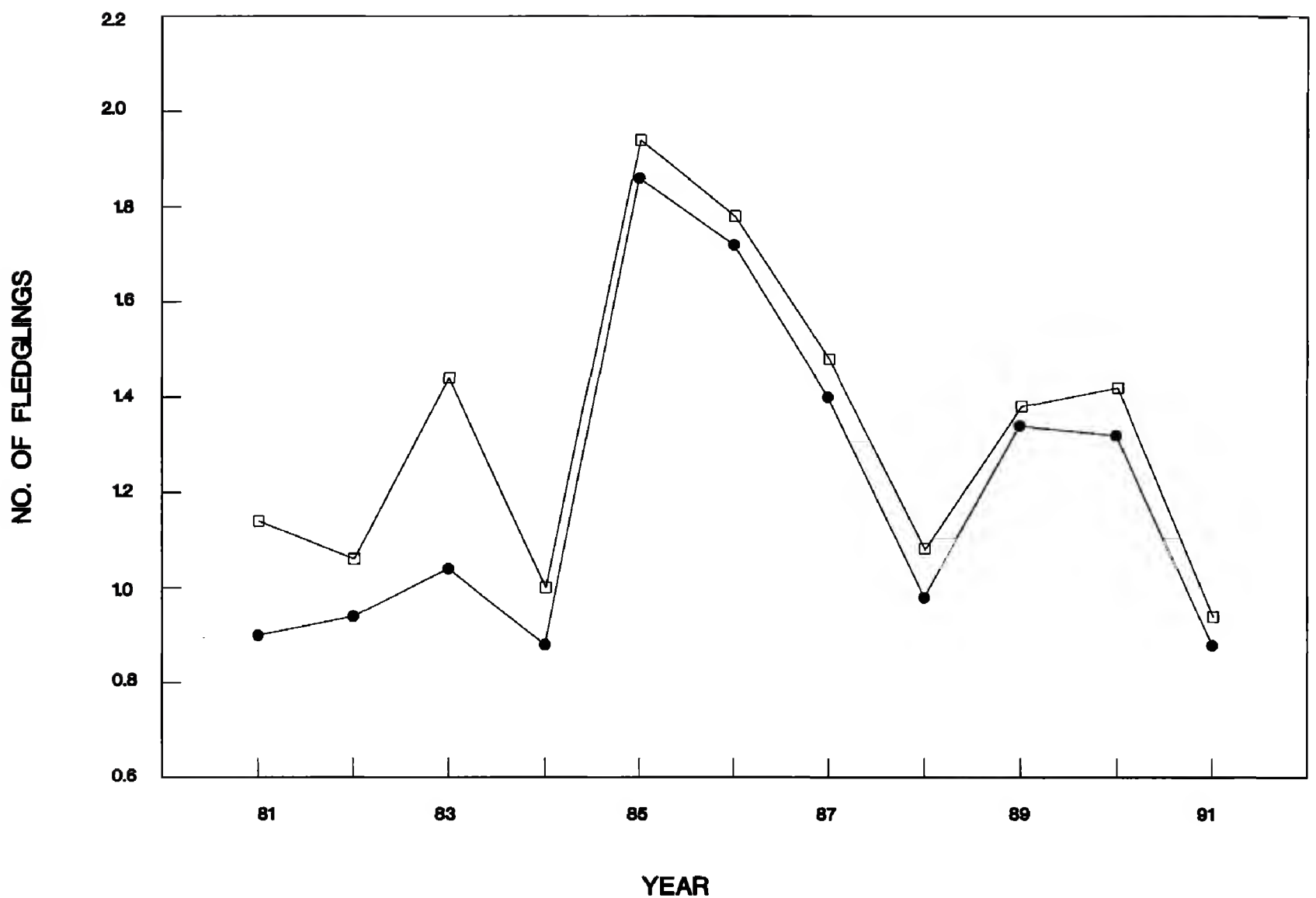


Figure 3. The mean reproductive rate for ospreys along the Umpqua River between 1981–91. Mean number of fledglings produced per occupied territory (solid circles) and the mean number of fledglings produced per breeding attempt (open squares).

occupied territories that were successful ranged from 42% in 1991 to 90% in 1985 (Fig. 3a) and the percent of breeding attempts that were successful ranged from 45% in 1991 to 93% in 1985. The number of fledglings produced per successful nest each year varied and was a function of the increasing population size and the highly variable reproductive rate (Fig. 3b).

The mean reproductive rate of ospreys using artificial platforms was 1.48 fledglings per breeding attempt (range = 0.86–2.22, $N = 99$). On naturally occurring substrates, the rate was 1.27 fledglings per breeding attempt (range = 0.85–1.83, $N = 204$). This difference in productivity between the two substrates was not significant ($t = 1.426$, $df = 301$, $P > 0.05$).

The mean reproductive rate of successful breeding attempts on artificial platforms was 2.21 fledglings per site (range = 1.50–2.57, $N = 66$) while on natural substrates, the rate was 2.05 fledglings per site (range = 1.40–2.43, $N = 127$). Here also,

analysis of the productivity from the two types of substrates indicated the difference was not significant ($t = 1.46$, $df = 191$, $P > 0.05$).

DISCUSSION

In Oregon, ospreys were thought to be rare (Gabrielson and Jewett 1940, Marshall 1969) until a mail survey by Roberts and Lind (1977) estimated a minimum population of 121 nesting pairs in 1971. The highest concentrations of birds were found at Crane Prairie Reservoir, Lookout Point Reservoir, and along the Rogue River. Henny et al. (1978), using ground and aerial surveys, estimated the number of nesting pairs in Oregon at 308 in 1976, with major concentrations at Crane Prairie Reservoir and the adjacent Deschutes National Forest, coastal lakes and reservoirs between Florence and North Bend, Rogue River, Lane County reservoirs, and the Umpqua River. These estimates may have been representative of the osprey population during the 1970s, but given the present level

of occupation along the Umpqua River, the current population levels in Oregon are probably much higher than the estimates made by Henny et al. (1978).

An historical example of the loss of habitat due to the draining and the associated impacts on ospreys was reported by Henny (1988) for Tule Lake in the Klamath Basin. He examined the historical field records and an unpublished manuscript and found that a very large osprey colony existed during the late nineteenth century at the northeast corner of Tule Lake along the Oregon border. He hypothesized that a radical decline in the osprey population occurred in the basin when construction work began on the U.S. Bureau of Reclamation's Klamath Project to drain Tule Lake in 1906.

More recently, Henny and Anthony (1989) reviewed the status and the reproductive performance of ospreys in the western states and found productivity usually ranged from 0.95–1.3 young per occupied territory and that organochlorine contaminants were still a problem during the 1980s for a few individual birds and in some localized areas. They concluded that recent population increases and range expansions were, in part, due to reduced DDE residues in the West.

The most likely explanation for the population increase during this study is a combined response by ospreys to the improved nesting habitat conditions along the river and to generally lower levels of pesticide contamination in the western United States (Henny and Anthony 1989).

The reproductive rates of the osprey varied considerably during the 11-yr period of the study. The coefficient of variation was 27.8% for the annual percent of occupied territories that were successful, 22.7% for the annual percent of breeding attempts that were successful, 29.0 % for the productivity of occupied territories, 24.4% for the productivity of breeding attempts, and 15.1% for the productivity of successful breeding attempts. When designing a long-term monitoring strategy for ospreys, consideration should be given to this variability in productivity and one should clearly expect years of low reproduction, even with healthy growing populations. Therefore, I would recommend sampling the population every three or four years to reduce the chance of only sampling low reproductive years, which may be weather dependent and cyclic in nature.

The rate of the population increase observed during this study was similar to the 64% increase

observed between 1966–72 on Fletcher Pond by Postupalsky (1978) and the 54% increase observed by Spitzer et al. (1983) from 1976–81 between New York City and Boston. In contrast, Rhodes (1972) observed an initial 160% increase in the population after installing only 12 artificial structures on an island refuge in Chesapeake Bay. The larger rate of increase was probably due to a smaller initial population (four to six nests on the ground) and to a virtual lack of suitable nesting habitat on the island. After the initial increase in the population the ospreys continued to increase but at a rate of about 38% over the next 3 years.

Unlike this study, the reproductive performance of ospreys has been shown to be greater on artificial than on natural substrates (Seymour and Bancroft 1983, Westall 1983). On Sanibel Island, Westall (1983) found production averaged 1.47 fledglings per breeding attempt on artificial structures and only 0.69 fledglings per breeding attempt on natural sites. In northeastern Nova Scotia, Seymour and Bancroft (1983) found mean production to be 1.29 fledglings per occupied nest on utility poles and 1.09 fledglings per occupied nest on natural sites. Similarly, Postupalsky (1978) found in Michigan that productivity was twice that recorded on natural sites. In contrast to these three studies, Rhodes (1977) observed during a 5-yr study on an island in Chesapeake Bay that productivity was 1.4 fledglings per breeding attempt on platform structures and 1.9 fledglings per breeding attempt at other sites (both natural and other man-made structures). The differing results from these studies may be related to the fact that they were not designed as controlled experiments and, therefore, influenced by several sources of bias (Postupalsky and Stackpole 1974).

During the 11 years of the study, platform occupancy rate (in terms of platform years) was 46%. Although lower than elsewhere, the rate was comparable to the 50–60% occupancy rates reported in California (Garber et al. 1974), Maryland (Reese 1977), and Michigan (Postupalsky 1978), but it was considerably lower than the 70% rate observed in Florida (Westall 1983) and the 78% rate found in the Chesapeake Bay (Rhodes 1977).

From a management perspective, the use of platforms and power poles along a river system clearly can be an effective tool for managing an expanding osprey population. Based on the fact that as much as 62.5% of the platform sites were occupied and pairs establishing themselves on platform sites

accounted for over 40% of the total population increase along the Umpqua River, the installation of artificial platforms has played an important role in contributing to the expansion of ospreys along this river.

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POSSIBLE SECOND CLUTCHES IN A MEDITERRANEAN MONTANE POPULATION OF THE EURASIAN KESTREL (*FALCO TINNUNCULUS*)

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ABSTRACT.—Three of eleven Eurasian kestrel (*Falco tinnunculus*) pairs from a montane population in central Spain began possible second clutches after chicks of the first had fledged. The mean-laying date of first clutches for double-brooded pairs was earlier than single-brooded pairs and double-brooded pairs laid larger first clutches and fledged more young than single-brooded pairs. Overall, second clutches were less productive. The low latitude and high availability of prey in this study area may explain the occurrence of the second breeding attempts in this species.

KEY WORDS: *Falco tinnunculus*; *Eurasian kestrel*; *second clutch*; *Mediterranean*.

RESUMEN.—Tres de once parejas de Cernícalo Vulgar (*Falco tinnunculus*) en una población montana de España central, iniciaron una presumible segunda puesta tras haber volado los pollos de la primera. Se observó un descenso de la productividad de los cernícalos a medida que avanzó la estación reproductora. La fecha de puesta de las consideradas parejas con dobles puestas fue más temprana que la de las parejas que sólo hicieron una. En las primeras puestas de las parejas que presumiblemente pusieron dos veces, el número de huevos y el de pollos volados fue mayor que en las que sólo hicieron una puesta. Las consideradas segundas puestas fueron menos productivas que el resto de las puestas. La baja latitud junto con la gran abundancia y constante disponibilidad de presas en nuestro área de estudio podría ser la causa de segundas puestas en esta especie.

[Traducción de Juan Fargallo]

Most diurnal raptors outside the tropics raise only one brood per year; however, in some species, some pairs occasionally raise two broods in the same year (Newton 1979). Second clutches have been noted in rodent-eating raptors, such as the American kestrel (*Falco sparverius*) (Toland 1985), white-tailed kite (*Elanus leucurus*) and black-shouldered kite (*E. caeruleus*) (Newton 1979). The breeding cycle of the Eurasian kestrel (*F. tinnunculus*) has been well documented in central and northern Europe (e.g., Cavé 1968, Newton 1979, Village 1990, Palokangas et al. 1992, Cramp & Simmons 1980). No long-term study of the breeding biology of this species has ever documented second clutches in breeding pairs (Cavé 1968, Meijer et al. 1990, Daan et al. 1990, Village 1990) and only

one case of a second brood in free-living kestrels has ever been recorded in Spain (Sánchez 1990). In captive kestrels, Meijer (1989) and Meijer et al. (1992) found second clutches under conditions of manipulated photoperiod. Here, we report three instances of possible second breeding attempts in the same year by Eurasian kestrels in a montane area of central Spain.

STUDY AREA AND METHODS

The study was conducted in the Campo Azávaro region of central Spain (40°40'N, 4°20'W), an extensive grassland area located on the northern slopes of Sierra de Malagón at 1300 m elevation. All nests were checked for occupancy by kestrels and laying dates, clutch sizes, and numbers of young hatched and fledged were recorded. No adult kestrels were banded or color-marked so

adults could not be individually identified. The possible double clutches we are reporting are based on the isolation of individual pairs within the study area and associated field observations that lead us to believe that second breeding attempts were being made by the same females. All nestlings in nest boxes were banded at 15–20 days of age.

RESULTS AND DISCUSSION

Of the 44 kestrel nest sites we observed in the study area in 1993 and 1994, 29 (66%) were in old carrion crow (*Corvus corone*) nests (20 in trees and 9 on metal utility structures), 9 (21%) were in holes of buildings, and 6 (13%) in nest boxes. Three breeding attempts in nest boxes (nests A, B and C) were presumed to be the second breeding attempts of females in the same season. In one case (nest A), the female attempted to breed a second time in the same nest box, and in the two other cases the females moved only 150 m away to the next available nest box. The closest neighboring, breeding females were 1.5–2.5 km away. American kestrels lay their second clutches either in alternative nest sites close to the original nest, or in the same nest if alternative sites are not available (Toland 1985). All three of these females were the first to initiate clutches in the study area. We could not individually identify these females but all three were observed delivering prey to banded fledglings that remained in family groups on top of the nest boxes and these same banded fledglings were observed standing on the tops of boxes while females incubated inside. Because female Eurasian kestrels are aggressive toward intruding conspecific females during the breeding season (Wiklund and Village 1992), it seemed unlikely that other females had both initiated breeding attempts and had adopted fledglings and associated parental feeding behavior.

The three clutches contained one (nest A), four (nest B) and three eggs (nest C), and all three clutches were incubated. Eggs hatched in only one clutch (nest C) which successfully fledged three young. The mean laying date of the first clutches of these three females (29 March) was significantly earlier than for females that attempted to breed only once (10 May) (Mann-Whitney test: $U = 2.35$, $N = 11$, $P = 0.01$) with a difference in median laying dates of 41 days. The laying date of the latest clutch laid by females breeding only once was 27 May. Laying dates of second clutches were 16 June (nest A), 12 June (nest B), and 8 June (nest C). These dates were

significantly later than females that laid only once (Mann-Whitney test: $U = 2.3$, $N = 11$, $P = 0.01$). Females laying twice had significantly larger first clutches (6, 6 and 6 eggs) than females that laid only once ($\bar{x} = 4.5 \pm 0.9$, $\pm SD$, $N = 8$; Mann-Whitney test: $U = 2.13$, $N = 11$, $P = 0.03$). The average number of chicks hatched per clutch was smaller in females that laid once ($\bar{x} = 4.0 \pm 1.4$, $\pm SD$, $N = 6$) than in the first broods of females that laid twice (6, 6 and 5), but the difference was only marginally significant (Mann-Whitney test: $U = 1.58$, $N = 9$, $P = 0.1$), probably due to the small sample size. Late clutches (1, 3, and 4) and broods (0, 3, 0) of females breeding twice were smaller than those of females breeding only once but, again, the difference was only marginally significant (Mann-Whitney test: $U = 1.79$, $N = 11$, $P = 0.07$, and $U = 1.83$, $N = 9$, $P = 0.06$, respectively). Females nesting twice averaged a greater number of young fledged (6, 6 and 5) from their first nesting attempts than females that bred only once ($\bar{x} = 3.8 \pm 1.1$, $\pm SD$, $N = 6$; Mann-Whitney test: $U = 1.98$, $N = 9$, $P = 0.04$). Likewise, fledging success of the first brood of females breeding twice was also higher (100, 100 and 83%) than the average fledging success of females that bred only once ($\bar{x} = 80.6 \pm 11.2$, $\pm SD$, $N = 6$). Again, this difference was only marginally significant (t -test: $t = 1.73$, $N = 9$, $P = 0.1$).

In species previously documented to breed more than once in a single season, an early initiation of breeding allows pairs to make a second breeding attempt, while single-brooded pairs delay the onset of breeding until conditions allow them to lay their optimal clutch size (Lack 1954, Klomp 1970, Perrins 1970). This is consistent with our finding that only those females we presumed to have bred twice laid clutches earlier than 1 May and no second clutches were laid by females whose first clutches were laid in May. Sánchez (1990) has also reported that in Spain, Eurasian kestrels typically lay their clutches in May.

In captive and wild, double-brooded American kestrels (Bird and Laguë 1982, Toland 1985) and Eurasian kestrels (Meijer 1989, Palokangas et al. 1992), there is a seasonal decline in clutch size with increasing laying date. When all clutches in our study were considered, we found a similar seasonal decline in clutch size ($r = -0.80$, $F = 22.43$, $df = 1, 10$, $R^2 = 65.1$, $P < 0.001$; Fig. 1). These data suggest there may be a seasonal decline in Eurasian kestrel productivity.

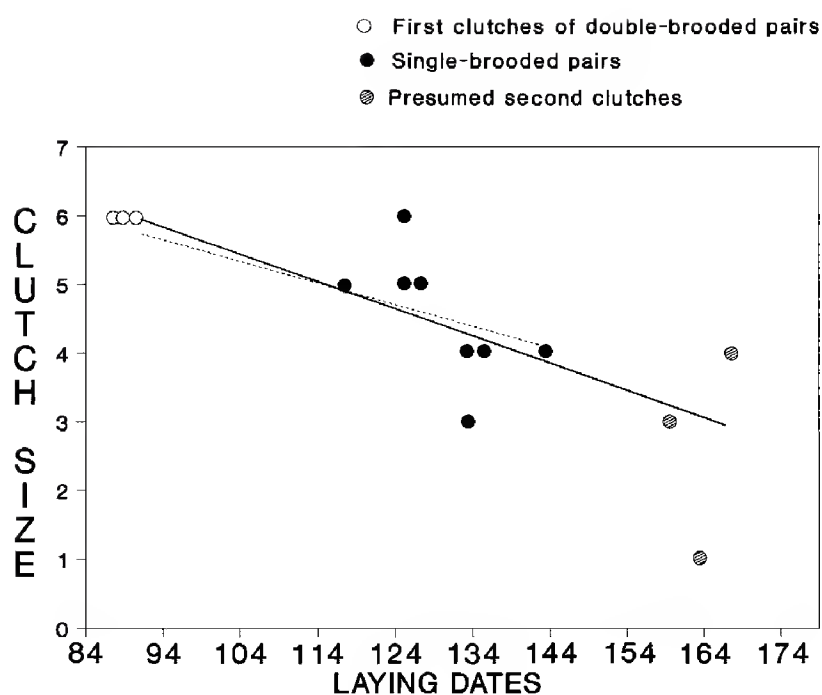


Figure 1. Relationship between clutch size and laying date (1 = 1 January) in clutches (first clutches only dashed line and second clutches only solid line) of Eurasian kestrels in Spain.

In birds of prey, the duration of the breeding season is associated with latitude. The proportion of replacement clutches of large diurnal raptors, with longer breeding cycles and of second clutches of small diurnal raptors with short breeding cycles, is less frequent in the northern portions of their ranges (Newton 1979). An inverse correlation between vole (*Microtus* spp.) abundance and the laying date of Eurasian kestrels has been well documented (Cavé 1968, Dijkstra et al. 1982), and there appears to be a shorter postfledging dependence period (16 days on average) in Eurasian kestrels in Mediterranean regions with a high abundance of prey (Bustamante 1994). We believe that the combination of latitudinal effects and high vole abundance in montane regions of the Mediterranean (Veiga 1982, 1985, 1986), allows an early initiation of breeding in Eurasian kestrels followed by a second breeding attempt in pairs with high-quality territories.

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MEXICAN SPOTTED OWL HABITAT CHARACTERISTICS IN ZION NATIONAL PARK

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ABSTRACT.—We studied Mexican spotted owl (*Strix occidentalis lucida*) distribution, density, and habitat characteristics in Zion National Park from 1989–1991. We found 28 owls (12 pairs and 4 single males) at 16 different locations throughout the park. Estimated crude density ranged from 0.018–0.042 owls/km² while estimated ecological density ranged from 0.216–0.738 owls/km² over 3 years. Owls were associated with narrow canyons that usually contained a water source. Spotted owls used canyons that had greater absolute humidity and more vegetation strata than canyons selected at random. The geomorphology of these canyons may provide conditions compensatory to the complex forest structure associated with the owl elsewhere within its range by providing cool roosts and nest sites.

KEY WORDS: *density; habitat; Mexican spotted owl; microclimate; Strix occidentalis lucida; Zion National Park.*

Características del hábitat de *Strix occidentalis lucida* en el Parque Nacional Zion

RESUMEN.—Entre 1989 y 1991, estudiamos la distribución, densidad y características del hábitat de *Strix occidentalis lucida* en el Parque Nacional Zion. Detectamos 28 individuos (12 parejas y cuatro machos), en 16 lugares diferentes del Parque. La densidad cruda estimada se encontró en el rango de 0.018–0.042 búhos/km², mientras que la densidad ecológica estimada se encontró en un rango de 0.216–0.738 búhos/km², en un período de tres años. Los búhos estaban asociados a estrechos cañones y que usualmente contenían agua. *Strix occidentalis lucida* usaban cañones que presentaban una gran humedad absoluta y más estratos vegetacionales que cañones seleccionados al azar. La geomorfología de estos cañones puede proporcionar condiciones compensatorias para la compleja estructura boscosa asociada con este búho en su rango de distribución, al proveer sitios de descanso fríos y de nidificación.

[Traducción de Ivan Lazo]

The Mexican spotted owl (*Strix occidentalis lucida*) occupies a broad geographic area in North America, but does not occur uniformly throughout its range (Gutiérrez et al. 1995). It occurs from the four corners states south to Michoacan, Mexico and reaches the northwestern limit of its range in southern Utah where the habitat is naturally fragmented. Disjunct canyon systems on the Colorado Plateau contain steep-walled canyons with little or no vegetation that provide unique habitat for spotted owls (e.g., Kertell 1977). This habitat is in stark contrast to forested canyon areas used by owls throughout the southwestern United States.

Spotted owls were reported from southern Utah as early as 1928 (Hayward et al. 1976). Prior to 1989, they were recorded in Glen Canyon National Recreation Area, Bookcliff Range, and Zion Na-

tional Park (Behle 1960, 1981, Kertell 1977). Zion National Park contained the majority of historical locations (Kertell 1977). Yet no habitat assessment was available for the canyon habitats of the Colorado Plateau geographic province at the time the Mexican spotted owl was listed as a threatened subspecies (USDI 1993). Therefore, we investigated the distribution, density, and habitat characteristics of Mexican spotted owls in this unique canyon habitat.

STUDY AREA

Our study area was Zion National Park (59,353 ha) in southwestern Utah (National Park Service 1987). The climate was characterized by hot, dry summers and mild winters (Krell 1988). Temperatures ranged from –12–40°C and annual precipitation ranged from 30–50 cm (National Park Service 1987). Elevations ranged from 1,109–2,660 m (Brereton and Dunaway 1988). The Park was dominated by sheer cliffs of Navajo sandstone, slick-rock terraces, and hanging canyons (e.g., a side canyon whose mouth lies above the floor of the main canyon). The terrain was extremely rugged with minimal access provided by roads and trails.

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Four vegetation communities dominated the Park: ponderosa pine (*Pinus ponderosa*), oak (*Quercus* spp.), sagebrush/pinyon-juniper (*Artemisia* spp./*P. monophylla*-*Juniperus* spp.), and riparian (National Park Service 1987). The ponderosa pine community was characterized by ponderosa pine, Douglas-fir (*Pseudotsuga menziesii*), and white fir (*Abies concolor*); the oak community consisted of Gambel oak (*Q. gambelli*) of a low-growing brushy stature, and scrub oak (*Q. turbinella*); the sagebrush/pinyon-juniper community was primarily sagebrush, pinyon pine, Utah juniper (*J. osteosperma*), and cacti (*Opuntia* spp.); riparian zones included species such as cottonwood (*Populus fremontii*), willow (*Salix* spp.), boxelder (*Acer negundo*), ash (*Fraxinus* spp.), and big-toothed maple (*A. grandidentatum*). Within the canyon riparian zones were "stringers" (vegetation arranged in long, thin strips) of ponderosa pine, white fir and Douglas-fir.

METHODS

During 1989–1991, we located owls using standard survey techniques (Forsman 1983, Franklin et al. 1990) with some modifications necessary to compensate for the rugged terrain (Rinkevich 1991). Modifications included calling from prominent ridge and mesa tops overlooking a canyon (e.g., point surveys) for 10–15 min at the top of each hour and then listening for the remainder of the hour during 3–5 hr night time periods. We also used a 45.7 cm diameter parabolic dish with headphones (Geleco Electronics Ltd., Ontario) to listen for owls calling in inaccessible canyons. We considered an owl location as a visual sighting of at least one adult spotted owl or a minimum of two auditory detections in the same canyon in the same year.

We surveyed the Park for owls using a stratified, random sampling scheme, using canyon and plateau areas as strata. These two areas were surveyed in proportion to their total area during 1989–90. In 1991, we surveyed as many canyons and forested areas as possible within the Park and resurveyed all canyons that were sampled in previous years.

We estimated crude density for each year by dividing the number of owls found in Zion Park by the size of the Park (Franklin et al. 1990). We also calculated ecological density by dividing the number of owls by the amount of suitable habitat (Franklin et al. 1990). We estimated suitable spotted owl habitat from the area of (1) canyons in which owls were detected during our surveys, (2) canyons that had been used by owls prior to our study (e.g., areas within the Park that had a previous record of spotted owls), and (3) canyons that shared similar characteristics to those in which we found owls. Because we intensively surveyed the Park for 3 yr, we were confident that our estimate of suitable habitat was reasonable.

We compared habitat characteristics of canyons where owls were found (owl canyons) with randomly-selected canyons which we surveyed, but in which we did not find owls (random canyons). We did not compare owl canyons to plateau areas because we did not find owls in plateau areas. Since some random locations could potentially harbor owls, this was a conservative test of the differences in habitat characteristics between used and available habitat.

Habitat sampling within owl canyons was accomplished

in three ways. First, we centered a sample plot directly below the observed roost position of an owl (Solis 1983). When owls were found roosting on cliffs, sample plots were placed directly below the owl as close to the cliff as possible. Thus, transect lines often were parallel to the canyon wall. Second, we measured habitat throughout the canyon using a stratified, random sampling scheme when we found an owl but were unable to directly observe the bird (see below). Third, we sampled habitat using a stratified, random sampling scheme in canyons where owls historically occurred. Because no known major habitat or geologic changes occurred in previously occupied habitat, we assumed that the absence of owls from these historic sites was a function of owl demography (e.g., LaHaye et al. 1994) rather than due to changes in habitat.

We used the same stratified, random sampling scheme to measure habitat characteristics within both random and owl canyons. We first proportionately allocated the number of sample plots according to canyon size (e.g., more plots were measured in longer canyons). Then, we randomly selected sample plots throughout these canyons (Rinkevich 1991). Since longer canyons contained more plots, we averaged all plots within a canyon to obtain a single mean value for each canyon. The mean values of random plots were compared to the mean values of plots within owl canyons.

We measured 43 habitat characteristics within owl and random canyons (see Rinkevich 1991 for a complete list). Of the 43 characteristics, we measured five geomorphic (e.g., canyon width, length), two microclimatic (e.g., temperature and humidity), and 35 vegetation (e.g., tree height, basal area, strata layer) variables. Vegetation sampling followed Bias and Gutiérrez (1992). We used a variable circular plot method with a 20 basal-area factor wedge prism to define sample trees (Mueller-Dombois and Ellenberg 1974, Dilworth 1981). Geomorphic features were measured using a tape measure, pacing with a measured stride, or estimated from topographic maps. Presence or absence of surface water was recorded. Ambient air temperature and percent relative humidity also were measured at each plot center. Relative humidity was measured using a sling psychrometer and later converted into absolute humidity (Ruskin 1965). Absolute humidity could then be analyzed independent of temperature. We assessed differences in habitat characteristics between owl and random canyons using Mann-Whitney *U* tests because the data were not normally distributed.

RESULTS

Density. We surveyed approximately 75% of Zion National Park within which we found 28 spotted owls (12 pairs and 4 single males) at 16 locations during 1989–91. Fourteen owl locations were in the Park and 2 were located on Bureau of Land Management (BLM) Wilderness Study Areas (WSAs) adjacent to Park boundaries. Because these latter two sites were within meters of the Park boundary they were included in density estimates. We detected 3 pairs and 3 single male owls at six

Table 1. Characteristics of canyons used by Mexican spotted owls and randomly sampled canyons in Zion National Park, Utah.

VARIABLE	OWL CANYONS MEAN (SD) (N = 13)	RANDOM CANYONS MEAN (SD) (N = 17)	MANN-WHITNEY U VALUE (P)
Absolute humidity ¹	9.41 (1.43)	6.51 (1.92)	24.0 (0.000)
Vegetation strata ²	2.79 (0.31)	2.29 (0.41)	47.5 (0.007)
Percent canopy closure	0.43 (0.14)	0.39 (0.30)	99.0 (0.630)
Max. shrub height ³	2.25 (35.80)	2.15 (29.53)	98.0 (0.601)
Min. shrub height ⁴	0.35 (9.87)	0.36 (9.32)	113.5 (0.899)
Mature tree BA ⁵	9.09 (12.65)	16.53 (26.44)	118.0 (0.716)
Medium tree BA ⁶	13.17 (12.56)	8.34 (10.99)	83.0 (0.226)
Total live BA ⁷	23.23 (21.47)	23.73 (24.93)	107.5 (0.899)
Snag BA ⁸	0.38 (1.39)	0.69 (1.96)	115.5 (0.688)
Large woody debris ⁹	0.63 (0.98)	0.59 (0.86)	104.0 (0.768)
Small woody debris ¹⁰	0.54 (0.76)	0.69 (1.34)	112.0 (0.95)
Temperature (°C)	21.6 (8.49)	20.0 (11.94)	93.0 (0.46)
Canyon width ¹¹	85.13 (52.97)	117.69 (67.70)	137.5 (0.258)
Ledge height ¹²	21.93 (31.20)	21.27 (44.59)	87.5 (0.296)
Bench height ¹³	352.6 (371.03)	279.80 (376.08)	94.0 (0.490)

¹ Absolute humidity (gm/cm³ of water).
² Number of vegetation layers (canopy, shrub, and herb) counted at each plot.
³ Maximum shrub height (m) of tallest shrub in plot.
⁴ Minimum shrub height (m) of smallest shrub in plot.
⁵ Basal area of mature size trees (m²/ha) (dbh from 52.5 to 89.8 cm).
⁶ Basal area of medium trees (m²/ha) (dbh from 27.5 to 52.4 cm).
⁷ Total basal area of all live trees in plot (m²/ha).
⁸ Total basal area of all snags in plot (m²/ha).
⁹ Percent of ground covered by large woody debris (>30.0 cm in diameter at large end).
¹⁰ Percent of ground covered by small woody debris (>2.5 to 30.0 cm in diameter at small end).
¹¹ Canyon width (m) at plot center using tape measure, pacing, or estimating from map.
¹² Height (m) to nearest ledge from canyon floor visually estimated.
¹³ Height (m) to first bench from canyon floor (measured visually or from topographic map).

locations, 7 pairs and 2 single males at 9 locations, and 12 pairs and 4 single males at 16 locations in 1989, 1990, and 1991, respectively. Estimated crude density ranged from 0.02 owls/km² in 1989 to 0.04 owls/km² in 1991 with a mean of 0.03 owls/km² (95% CI = 0.018–0.042) for the 3 yr sampling period. Ecological density ranged from 0.26 owls/km² in 1989 to 0.71 owls/km² in 1991 with a mean of 0.48 (95% CI = 0.216–0.738).

Habitat. All spotted owls were found in narrow, steep-walled canyons (Table 1). No owls were located on plateaus or mesas, although some owls flew onto canyon rims in response to our calling. Elevations of owl canyons ranged from 1,277–2,000 m. Owls used trees, cliffs or rock ledges as roosts in these canyons. Of the 16 canyons in which owls were found, 8 (50%) had perennial streams, 6 (37.5%) had ephemeral water sources, and 2 (12.5%) were inaccessible and, thus, avail-

ability of water was unknown. Of the 17 random canyons in which we did not find owls, 2 (11.5%) had perennial streams, 3 (17.5%) had ephemeral water sources, and 12 (71%) had no water present.

Of the 16 owl canyons, 2 canyons were inaccessible and 3 were found late in the study so no habitat data were collected for them. The remaining 11 owl canyons plus 2 historical canyons were used in our analysis. We sampled 54 habitat plots in the 13 canyons. We sampled 17 random canyons (44 habitat plots) for comparative purposes.

Owl canyons were very narrow and deep and contained limited but structurally diverse vegetation (Table 1). Random canyons were similar to owl canyons in most respects, but there were more vegetation strata in owl canyons than in random canyons. In addition, the combination of more free water and more vegetation strata in owl can-

yons may have contributed to the higher absolute humidity measured in the owl canyons.

DISCUSSION

Density. Our estimated crude densities of spotted owls in Zion National Park were lower than any published record (Franklin et al. 1990, Gutiérrez and Pritchard 1990, Bias and Gutiérrez 1992) for spotted owl populations. We found more owls in each year of the study primarily because of an increase in sampling efficiency rather than because of a true demographic change (e.g., annual increases were the result of sampling new areas). Thus, with greater sampling effort we would expect more owls to be found in Zion Park.

On the other hand, our estimates of ecological density were similar to density estimates for other owl populations. This suggested that the number of owls in Zion National Park may be related more to the availability of specific canyon habitats rather than to demographic processes per se. However, the lack of striking habitat differences between owl and random canyons suggests that there may be habitat at Zion that is currently unoccupied. While our sample population was small, it represented approximately 40% of the known owls within southern Utah.

Habitat. Spotted owls have been considered dependent on forests with complex structure (Forsman et al. 1984, Chávez-León 1989, Call 1990, Bias and Gutiérrez 1992, Verner et al. 1992, Gutiérrez et al. 1995). Very little typical spotted owl habitat occurs on the Colorado Plateau, and almost none in Zion National Park. However, the geomorphic relief apparently provides suitable habitat for spotted owls possibly by modifying microclimate and providing habitat structure. Owl habitat within canyons elsewhere in southern Utah ranges from rocky canyons containing patchy vegetation to narrow canyons containing little or no vegetation (pers. obs.).

Most of the owls we found were located in inaccessible, hanging canyons or steep-walled, narrow canyons and not in broad canyons with extensive sun exposure. Kertell (1977) also reported spotted owls associated with this type of habitat in Zion Park. The association of spotted owls with steep canyons has been reported from New Mexico (Johnson and Johnson 1985, Skaggs and Raitt 1988, Seamans and Gutiérrez 1995) and Arizona (Ganey and Balda 1989). However, the geomor-

phic relief within these areas did not approach that of Zion (pers. obs.).

Although Mexican spotted owls use rocky canyon habitat throughout their range it represents a relatively small proportion of spotted owl habitat in the southwest (USDI 1993, 1995). This suggests that only a limited subset of canyons in the subspecies' range contain the characteristics that provide suitable owl habitat.

In contiguous forests, spotted owls often have overlapping home ranges (Solis 1983, Forsman et al. 1984). The isolated nature of the deeply incised and extremely narrow canyons in the Park suggests that many owls in Zion have nonoverlapping home ranges because they are separated structurally and acoustically from adjacent canyons.

Mexican spotted owls appear to use canyon habitat because the geological and vegetation features produce distinctive environmental conditions important to spotted owls. Spotted owls respond to warmer temperatures by seeking cooler microclimates for roosting (Barrows and Barrows 1978, Solis 1983, Forsman et al. 1984). They have a narrow thermal neutral zone and experience heat stress at relatively low temperatures (Ganey et al. 1993). Thus, selection of cool, multi-layered forests in warm climates may be partly a response to physiological stress. It appears that the narrow canyons of Zion also modify local temperatures and humidity, particularly if water is available (see also Forsman 1976, Barrows 1981). In addition, the microclimate in these canyons may allow the development of forests with more complex structure, which are associated with spotted owls throughout most of their range. The more developed vegetation in owl canyons may also be advantageous to the owls' small mammal prey.

It appears that the geomorphology of the canyons provides roosting and nest sites for owls, modifies microclimate favorably, and allows more structurally diverse vegetation to develop. The vegetation in turn provides roost sites and possibly more habitat and food for the owls' prey. In these ways the unique features of these rugged canyons facilitate occupation by owls in an otherwise inhospitable landscape.

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FLEDGING AND MIGRATION OF JUVENILE BALD EAGLES FROM GLACIER NATIONAL PARK, MONTANA

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ABSTRACT.—During 1985–95, we documented fledging, migration, and subsequent locations of juvenile bald eagles (*Haliaeetus leucocephalus*) from Glacier National Park (GNP), Montana. The median fledging date was 1 August ($N = 29$). We radiotagged 11 fledglings, nine of which also received wing markers. The median date of migration from natal areas was 13 September ($N = 15$). The interval between fledging and migration varied from 32 to 70 d (median = 42 d, $N = 15$). Juveniles appeared to migrate alone, joining other eagles at foraging sites. GNP adults remained on their nesting territories when juveniles departed. One juvenile wintered 130 km from GNP. Others migrated as far as 1000 km. Six migrated to southern Montana, Idaho, Wyoming, and California. Three moved west to Washington or British Columbia. Two juveniles from the 1988 Lake McDonald nest migrated separately to the Pacific Coast. By 1991, one Lake McDonald adult had been replaced; the juvenile produced that year migrated south to Idaho. This contrast suggests that juveniles inherited distinct migration direction “programs” from different parents. Early autumn migration departures of GNP juveniles also may be genetically determined; we found no evidence that they remained locally to feed on autumn spawning runs of kokanee salmon (*Oncorhynchus nerka*) in GNP. At least 10 of the 11 radio-tagged juveniles survived their first winter. During spring migration, four juveniles passed through or near GNP. Nine summering sites or last known spring locations were in Alberta or British Columbia, Canada. There is no evidence to date of marked juveniles returning to breed in GNP natal areas.

KEY WORDS: *Haliaeetus leucocephalus*; bald eagle; fledging; Glacier National Park, Montana; migration; telemetry.

Volantones y migración de individuos juveniles de *Haliaeetus leucocephalus* desde el Glacier National Park, Montana

RESUMEN.—Durante 1985 a 1995, documentamos etapas de volantón, migración y subsecuentes localizaciones de individuos juveniles de la especie *Haliaeetus leucocephalus* desde el Glacier National Park (GNP), Montana. La mediana de la fecha de volanteo fue el primero de agosto ($N = 29$). Radiomarcamos 11 volantones, nueve de los cuales recibieron marcadores de alas. La mediana de la fecha de migración desde las áreas natales fue el 13 de septiembre ($N = 15$). El intervalo entre la etapa de volantón y la posterior migración varía entre 32 a 70 días (mediana = 42 d, $N = 15$). Los juveniles parecen migrar solitariamente, uniéndose a otras águilas en los sitios de forrajeo. Los adultos del GNP permanecen en sus territorios reproductivos cuando los juveniles parten. Un juvenil inverna a 130 km del GNP. Otros migraron a una distancia de 1000 km. Seis migraron al sur de Montana, Idaho, Wyoming y California. Tres se movieron al oeste de Washington o de British Columbia. Dos juveniles del nido Lake McDonald 1988, migraron separadamente a la Costa del Pacífico. En 1991, un adulto de Lake McDonald ha sido reemplazado; el juvenil producido ese año migró al sur de Idaho. Este contraste sugiere que los juveniles heredaron distintos “programas” de dirección migratoria, desde los distintos padres. El comienzo de la migración, a principios del otoño, de juveniles del GNP también puede ser determinada genéticamente. No encontramos evidencia sobre su permanencia en el sitio mientras *On-*

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corhynchus nerka desarrollaba su carrera otoñal de desove en el GNP. Al menos 10 de los 11 juveniles radiomarcados sobrevivieron a su primer invierno. Durante la migración de primavera, cuatro juveniles pasaron a través o cerca del GNP. Nueve sitios de verano o las últimas localizaciones de primavera conocidas fueron en Alberta o British Columbia, Canadá. No hay evidencias como fechas de retorno reproductivo de juveniles marcados en áreas natales del GNP.

[Traducción de Ivan Lazo]

Our study began during long-term research (started in 1965) on bald eagles (*Haliaeetus leucocephalus*) at autumn concentrations in Glacier National Park (GNP), Montana (McClelland et al. 1982). Migrating bald eagles congregated at kokanee salmon (*Oncorhynchus nerka*) spawning runs in Lower McDonald Creek (LMC) in GNP during each autumn between 1939 and 1988. The peak count of eagles was 639 in 1981. Although 201 juvenile bald eagles were captured and banded in the migration phase of that study, between 1977–88 (McClelland et al. 1994), their natal areas were unknown. Migration of juvenile bald eagles has been documented from Saskatchewan by Gerrard et al. (1974, 1978) and Harmata et al. (1985), the Greater Yellowstone Ecosystem (GYE) of Wyoming, Idaho, and Montana by Swenson et al. (1986) and Harmata and Oakleaf (1992), California by Hunt et al. (1992a), and Texas by Mabie et al. (1994), but there was no information on dispersal and survival of juvenile eagles from GNP nests. Such data will increase knowledge of population relationships in these mobile birds and aid managers concerned with protecting resources used by migrating eagles.

Our objectives in this study included documenting fledging dates, tracking juvenile migration, and determining if GNP juveniles participated in autumn concentrations of eagles at LMC. The latter objective was partially compromised when the salmon population, which had often exceeded 100,000 spawners in LMC, began to collapse in 1986 (Spencer et al. 1991). By 1991, no salmon were observed in LMC.

STUDY AREA

Our primary study area was in and adjacent to GNP, in northwestern Montana (48°30'N, 114°00'W), although radiotracking took us into other western states and Canada. The northern boundary of GNP coincides with the Canadian border. The Continental Divide roughly bisects the Park, north to south. Juvenile bald eagles marked in our study were from five nests in GNP, a nest at Hungry Horse Reservoir (20 km southwest of GNP), and a nest at Cyclone Lake (4 km west of GNP). Nests were within 300 m of a lake. Lake elevations above sea level ranged

from 961 m at Lake McDonald to 1366 m at Saint Mary Lake.

METHODS

During occupancy and incubation periods, we monitored nests intermittently from fixed-wing aircraft or from the ground. Precise hatching dates were not determined. Near the expected fledging (departure from the nest) time, observations were made on a daily basis until fledging occurred. Eagles that fledged prematurely (before being capable of self-sustained flight) were captured and marked on the ground. Others were captured in natal territories using padded leg-hold traps placed in shallow water and baited with fish, 3 to 4 wk after normal fledging.

Captured eagles were banded with U.S. Fish and Wildlife Service aluminum leg bands. Eleven fledglings received backpack transmitters (model 200, Telonics, Inc., Mesa, AZ) weighing about 54 g; batteries were expected to operate for a minimum of 15 mo. This enabled tracking some eagles during their second autumn migration. Nine fledglings also were fitted with orange patagial wing markers with black, alpha-numeric codes. Additional details on capture technique, markers, transmitters, and method of sex determination were presented in McClelland et al. (1994).

Local movements and migration routes were tracked from the ground whenever possible. When transmitter signals were lost, tracking resumed in fixed-wing aircraft, usually a Cessna 182. We used telemetry not only to document locations, but also to lead us to sites where marked eagles were observed from the ground. Harmata (1984) and others also have used this approach. We used aircraft only when necessary to relocate eagles, and we tried to avoid low-level flights over national parks and other ecologically sensitive areas. Tracking ended when ground searches for the signal were unsuccessful and weather or lack of funds prevented flights. Local winter and summer movements of three juveniles were documented by volunteers. In spring, we monitored for transmitter signals in the GNP area and on trips in various parts of northwestern Montana. When a signal was found, we tracked the eagle as long as funds were available. Sighting reports were used only if the wing marker code was read.

RESULTS AND DISCUSSION

Fledging. Fledging occurred between 14 July and 1 September (median = 1 August, $N = 29$). Gerrard et al. (1974) reported that most young eagles ($N = 14$) in their Saskatchewan study area (55°24'N) fledged in the second and third week of August. In the GYE (42°51' to 45°25'N), mean fledging dates ($N = 35$) were as early as 7 July in

Table 1. Fledging, marking, and migration dates of 15 juvenile bald eagles from nests in and adjacent to Glacier National Park, Montana.

EAGLE NUMBER	SEX	YEAR	NEST LOCATION	FLEDGE DATE (HOUR)	MARK DATE	MIGRATE DATE (HOUR)	FLEDGE TO MIGRATE (DAYS)
01 ^a	M	1985	Hungry Horse Res- ervoir	7 Aug ^c	8 Aug	26 Sep	50
02 ^a	F	1986	Hungry Horse Res- ervoir	27 Jul (1347)	21 Aug	5 Oct	70
A-07 ^b	M	1986	Logging Lake	29 Jul (1358) ^c	31 Jul	20 Sep (1942)	53
A-08	M	1986	Logging Lake	1 Aug (1140)	3 Sep	6 Sep (1242)	36
A-05	M	1988	Logging Lake	21 Jul (0540)	8 Aug	30 Aug (1300)	40
A-06	F	1988	Logging Lake	23 Jul (1115)	18 Aug	9 Sep	48
A-09	M	1988	Cyclone Lake	14 Jul ^c	14 Jul	22 Aug	39
A-52	M	1988	Lake McDonald	9 Aug	6 Sep	24 Sep (1230)	46
A-95	M	1988	Lake McDonald	31 Jul (0700)	22 Aug	1 Sep (1430)	32
A-14	F	1989	Bowman Lake	22 Aug (1046) ^c	25 Aug	3 Oct (1700)	42
A-93	M	1991	Lake McDonald	30 Jul (0657)	30 Aug	9 Sep (1645)	41
— ^d		1987	Waterton Lake	4 Aug (1120)	—	13 Sep (1030)	40
— ^d		1989	Cyclone Lake	27 Jul	—	16 Sep	51
— ^d		1991	Logging Lake	1 Aug (1730)	—	6 Sep (0730)	36
— ^d		1992	Saint Mary Lake	31 Jul (1000)	—	5 Oct	66

^a Eagles 01 and 02 had black and white leg bands and transmitters, but not wing markers.
^b Eagles with wing markers are identified by the marker's alphanumeric code ("A" followed by two digits).
^c Eaglet fledged prematurely, prior to the time it could sustain flight.
^d Fledgling not captured, banded, or marked.

four population units (Swenson et al. 1986). Although we expected fledging to occur during afternoon southwest winds that typically develop in the study area, eight of 11 (ultimately radio-tagged) occurred before 1200 H (median = 1140 H); one occurred at 0540 H (eagle 05) with no wind (Table 1). We captured and marked fledglings between 14 July and 6 September (median = 21 August, *N* = 11). Four eaglets fledged prematurely. Eagle 09 flew to perch trees after 2 d on the ground. Eagles 01 and 14 made their first sustained flights 5 d after premature fledging. By the end of day 5, they could maneuver in the forest canopy and returned to their nests.

Eagle 07's premature fledging at Logging Lake in 1986 involved an unusual sequence of events. Winds blew the eaglet, estimated to be 10.5-wk old, from the nest on 29 July. It remained in dense foliage and received no food from the adults for 2 d. We brought the bird to the ground, made measurements, and equipped it with a transmitter. Its wing and tail feathers were 75% emerged. By 2 August, it had moved 0.5 km from the nest to a 45% slope under a forest canopy impenetrable to the adults. To prevent starvation, we carried the fledgling about 700 m, to the lake inlet. It was unable to fly for 2 wk. Often, when an adult landed on a shoreline perch, the fledgling ran down the shore-

Table 2. Telemetry and sighting locations of eight radio-tagged juvenile bald eagles during autumn and winter after departure from nesting territories in and adjacent to Glacier National Park, Montana. Distances in parentheses are for more than one day. Last known distances from nests are in bold. Locations on dates after short-distance movements are not listed. Locations for Eagle 14 are not shown; it moved south only 132 km, to Flathead Lake (see Fig. 3).

EAGLE NUMBER, YEAR MARKED, AND OBSERVATION DATES	LOCATION (NORTH LATITUDE WEST LONGITUDE)	km MOVED
Eagle 01 (1985)	Hungry Horse Reservoir, MT (48°14' 113°50') ^c	Nest
26–27 Sep 1985	Placid Lake, MT (47°07' 113°31') ^c	126
29 Sep ^a	Near Silver Star, MT (45°42' 112°18')	(184)
10 Oct ^a	10 km N Dillon, MT (45°17' 112°45')	(60)– 339
Eagle 02 (1986)	Hungry Horse Reservoir, MT (48°14' 113°50') ^c	Nest
5–7 Oct 1986	Meadow Creek, South Fork Flathead River (47°52' 113°26')	52
8 Oct ^a	Near Anaconda-Pintlar Mtns., SW Philipsburg, MT (46°03' 113°38')	200
9 Oct	Near Bannock Pass, MT/ID border (44°48' 113°19')	143– 384
Eagle 07 (1986)	Logging Lake, Glacier National Park, MT (48°46' 114°01') ^c	Nest
20 Sep 1986	Firefighter Mtn., Hungry Horse Reservoir, MT (48°19' 113°52') ^c	53
22 Sep ^a	Greenhorn Mountain, W Helena, MT (46°43' 112°17')	(212)
23 Sep	10 km N Ashton, ID (44°12' 111°31') ^c	289
25 Sep–1 Oct ^a	Teton Pass, WY (43°29' 110°59') ^c	(101)– 561
Eagle 07		
19 Oct 1987	07's second fall; first located SW Kalispell, MT (48°11' 114°29')	soaring
21–22 Oct ^a	Along Clark Fork River, near St. Regis, MT (47°18' 115°05') ^c	(109)
23 Oct	Near Selway River, E Lowell, ID (46°06' 115°16')	134
26 Oct ^a	W "He Devil Mountain," OR (45°19' 116°41')	(163)
3 Nov	Near Malheur River, N Juntura, OR (43°46' 118°06')	(207)
5 Nov	8 km E Upper Alkali Lake, CA (41°40' 120°01')	(233)
7–8 Nov	28 km NE Mount Observation, NV (40°55' 119°56')	(105)
9 Nov	Soaring 15 km SSE from Mount Observation, CA (40°39' 120°05')	27– 1 021
Eagle 08 (1986)	Logging Lake, Glacier National Park, MT (48°46' 114°01')	Nest
20 Jan 1990 ^b	S Fork Boise River, below Anderson Dam, ID (43°22' 115°32') ^d	600
Eagle 09 (1988)	Cyclone Lake, MT (48°42' 114°18')	Nest
6 Jan 1989 ^a	Lake Roosevelt, near Kettle Falls, WA (48°34' 118°05') ^{c,d}	222
Eagle 52 (1988)	Lake McDonald, Glacier National Park, MT (48°38' 113°52') ^c	Nest
24–27 Sep 1988 ^a	21 km S Canada border, Koocanusa Reservoir, MT (48°49' 115°09') ^c	103
28 Sep	10 km N Sandpoint, ID (48°19' 116°34') ^c	110
29 Sep ^a	Selkirk Mountains, 16 km E Lake Roosevelt, WA (48°07' 118°01') ^c	110
30 Sep	Columbia River, 32 km NW Grand Coulee, WA (48°04' 119°22') ^c	101
1 Oct ^a	5 km NW Darrington, WA (48°16' 121°39') ^c	171
11–15 Oct	S end Skagit River Delta, WA (48°18' 122°24')	(131)
2 Dec–29 Jan ^a	Near Stillaguamish River, near Arlington, WA (48°10' 122°03') ^c	26– 608
Eagle 95 (1988)	Lake McDonald, Glacier National Park, MT (48°38' 113°52') ^c	Nest
2–3 Sep 1988 ^a	Koocanusa Reservoir, U.S./Canada border (49°01' 115°10') ^c	(109)
4 Sep ^a	Near Dewar Creek, 42 km W Kimberly, BC, Canada (49°39' 116°34') ^c	124
5 Sep	Near Trout Lake, BC, Canada (50°35' 117°24')	123
6 Sep	Columbia River, 7 km N Revelstoke, BC, Canada (51°02' 118°13') ^c	77
7 Sep	Adams River at Shuswap Lake, BC, Canada (50°54' 119°34') ^c	97
8–10 Sep ^a	Nicola Lake, BC, Canada (50°13' 120°30') ^c	112
11 Sep	Fraser River, 3 km W Hope, BC, Canada (49°22' 121°29') ^c	118
13–15 Sep	Squamish River, 55 km N Vancouver, BC, Canada (49°44' 123°09') ^c	(136)
16 Sep	Mountains W Squamish River, BC, Canada (49°53' 123°20') ^c	(154)– 685

Table 2. Continued.

EAGLE NUMBER, YEAR MARKED, AND OBSERVATION DATES	LOCATION (NORTH LATITUDE WEST LONGITUDE)	km MOVED
Eagle 93 (1991)	Lake McDonald, Glacier National Park, MT (48°38' 113°52') ^c	Nest
10 Sep 1991	29 km WSW Pincher Creek, AB, Canada (49°26' 114°18')	(100)
11 Sep	Chain Lakes, AB, Canada (50°15' 114°15')	120
12 Sep	Soaring at W edge Calgary, AB, Canada (51°06' 114°15')	97
13 Sep ^a	E Bow Valley Provincial Park, AB, Canada (51°07' 114°58') ^c	58
14 Sep ^a	Soaring up Elbow River, AB, Canada (50°44' 114°51')	51
17 Sep	Near Tally Lake, MT (48°24' 114°35')	(274)
18 Sep ^a	S Swan Lake, Swan Valley, MT (47°53' 113°51') ^c	101
21–29 Sep	Lake McDonald, Glacier National Park, MT (48°38' 113°52') ^c	95
1 Oct	SE White Sulphur Springs, MT (46°27' 110°50')	(356)
3 Oct ^a	Hyalite Canyon, S Bozeman, MT (45°27' 110°57')	(152)
4 Oct	W Madison Jct., Yellowstone National Park, WY (44°39' 110°56')	90
5–19 Oct ^a	Near Henrys Lake outlet, ID (44°36' 111°23') ^c	37
22–29 Oct	Harriman State Park, Island Park Reservoir, ID (44°23' 111°28') ^c	26
1 Nov	Along Henrys Fork River, W Ashton, ID (44°04' 111°30') ^c	35–555

^a Sighting and telemetry location on this date.

^b Sighting only.

^c Roost location.

^d Autumn migration routes unknown.

line toward the adult, vocalizing continuously. During the first week, we occasionally placed fish on the lake shore, out of the juvenile's view, but at a location toward which it was moving. It found and ate most of the fish we left. Seven d after the premature fledging, the adults began to make prey deliveries along the lake shore. The fledgling's first sustained flight was on 12 August and on 19 August it returned to the nest. On 3 September we recaptured the fledgling, remeasured it, and fitted it with a wing marker (code A-07). Since its initial capture 34 d earlier, its weight had increased by 29% (from 4.2 to 5.4 kg). All feathers were fully emerged, increasing the wingspan by 25 cm and tail length by 8.5 cm. Ultimately, eagle 07 provided information during the subsequent two autumn migrations.

Before migration, all fledglings generally remained within 1 km of their nests and appeared to be totally dependent on adults for food. In the immediate postfledging period, adults often delivered food to the empty nest, after which the fledglings flew to it to eat. Wood (1992) also described bald eagle fledgling dependence on adults in Florida. Alonso et al.'s (1987:212) description of the relationship between adult and juvenile Spanish imperial eagles (*Aquila adalberti*) appropriately de-

scribes what we also observed: "As the young got older, the parents spent less time in their vicinity. Young were not seen hunting, but depended upon their parents for food. They begged and chased their parents throughout the postfledging period, with higher intensity at the end. Nevertheless, the adults became progressively more reluctant to feed them. . . ."

Initiation of Migration. Juveniles began migration from natal territories between 22 August and 5 October (median = 9 September, *N* = 15, Table 1). Departure time varied from 0730–1942 H, but seven occurred after 1200 H (median = 1300 H, *N* = 9). The interval between fledging and migration varied from 32 to 70 d (median = 42 d, \bar{x} = 46 d, *SD* = 9.0, *N* = 15). The mean interval for the four prematurely fledged eaglets also was 46 d. Mean intervals in other studies were 24 d in California (Hunt et al. 1992a), 49 d in Maine and Florida (McCollough 1986, Wood 1992), and 52 d in Saskatchewan (Gerrard et al. 1974).

Autumn Migration Routes and Wintering Areas. We documented autumn migration routes and/or wintering areas for nine juveniles (Table 2). Six juveniles moved primarily south and three moved predominantly west on their first migrations (Fig. 1). The southward routes were similar to those

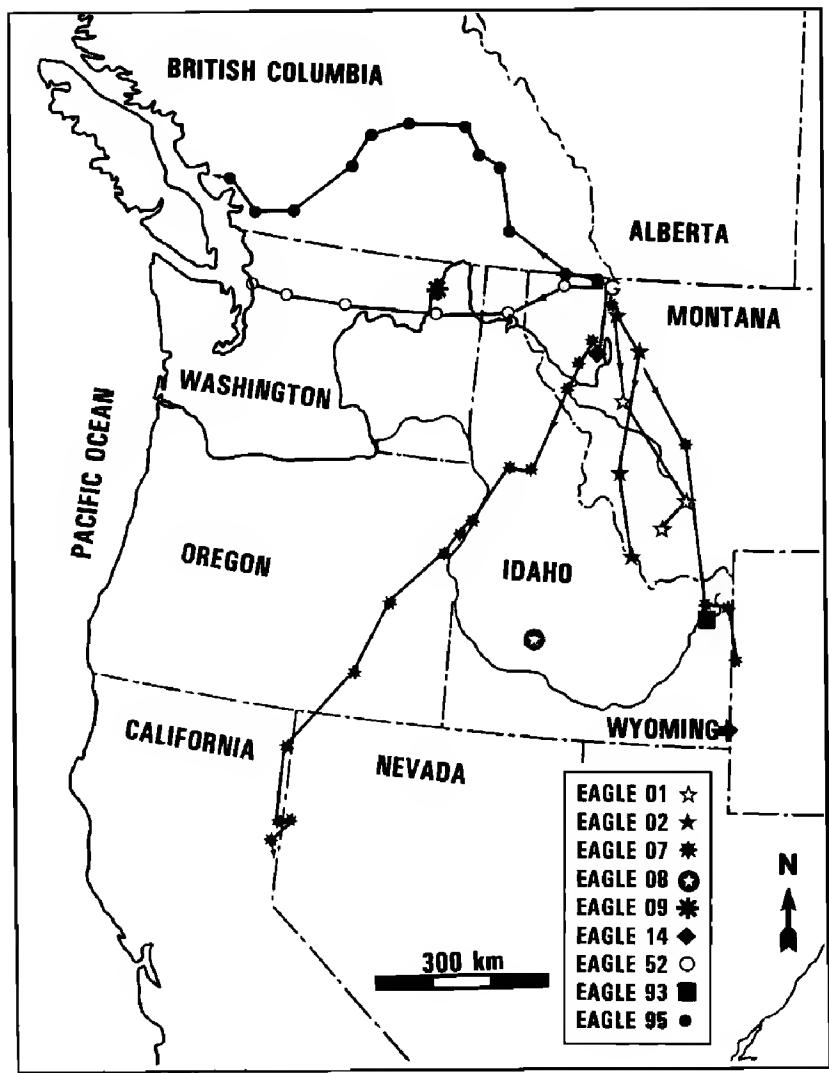


Figure 1. Autumn migration routes and last known autumn or wintering locations of bald eagles from natal areas in and adjacent to Glacier National Park (G symbol), Montana. Two routes are shown for eagle 07; the route farthest east is autumn 1986 (juvenile year) and the route farther west is autumn 1987 (second year). Migration routes for eagles 08 and 09 (wintering areas shown) were unknown. A more detailed route for eagle 14 is shown in Fig. 2. The migration route of eagle 93 is shown separately in Fig. 3.

of juveniles previously radiotracked from autumn concentrations in GNP (McClelland et al. 1994). Only eagle 14 remained within the general vicinity of GNP, wintering near Flathead Lake, 132 km south of its natal area (Fig. 2).

Eagle 09's route west from GNP was unknown, but it was found at Lake Roosevelt during a mid-winter waterfowl census of northeastern Washington (W.R. Radke, U.S. Fish and Wildl. Serv., pers. comm.). In the Kettle Falls District, where eagle 09 wintered, 98 bald eagles were observed on the 1989 winter count (G. LeBret, Natl. Park Serv., pers. comm.). Eagles 52 and 95, both from the 1988 Lake McDonald nest, were tracked west to Pacific coastal waters in Washington and British Columbia. Eagle 95 followed a curving path

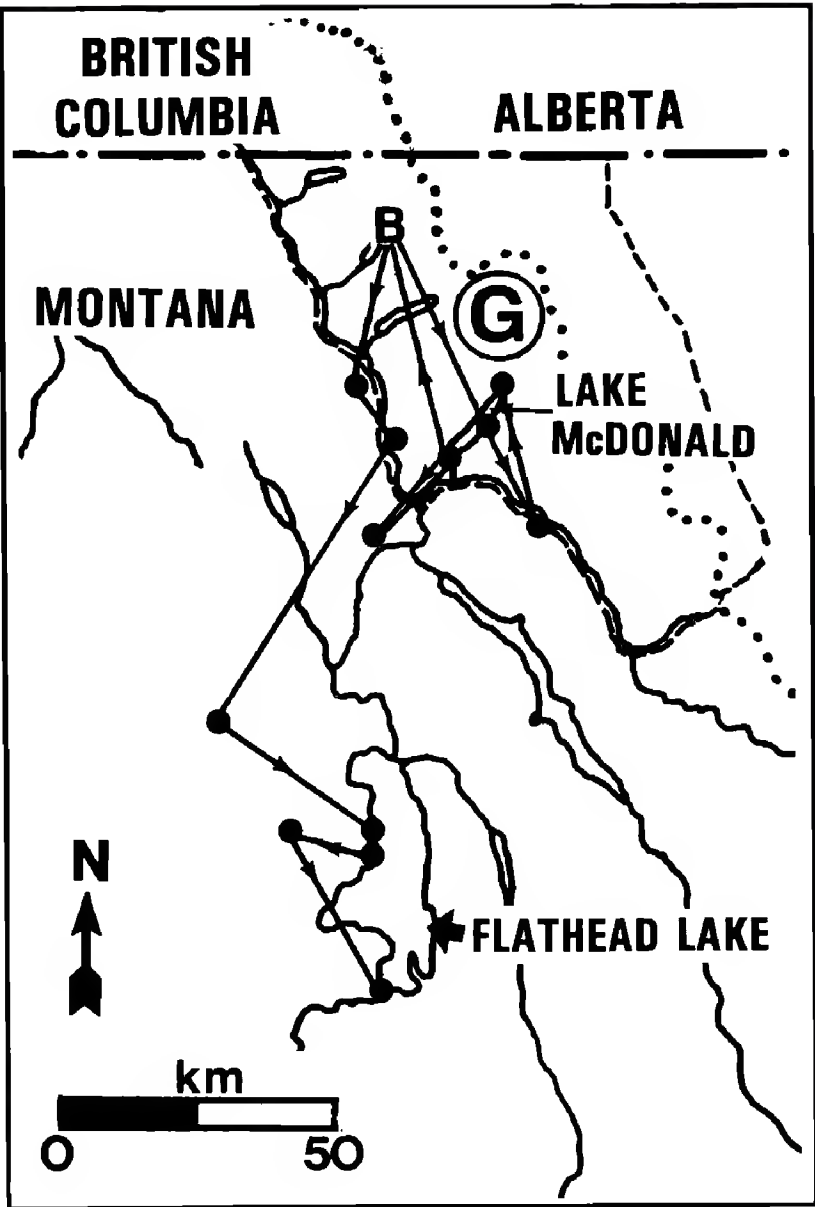


Figure 2. Autumn movements of juvenile eagle 14 in 1989, from the natal area at Bowman Lake (B symbol), in Glacier National Park (G symbol with boundary shown as dashed line), Montana, to wintering location at and near Flathead Lake, Montana. Note eagle 14's passage through the former kokanee salmon spawning area at Lake McDonald. Dotted line identifies the Continental Divide.

through British Columbia. During one period, we followed eagle 95's signal on the ground for 380 km through British Columbia, making frequent sightings. Eagle 52 took a relatively direct route, south of the Canadian border, to western Washington (Fig. 1) and it remained in the Skagit and Stillaguamish River areas throughout the winter. On some days, its transmitter signal was monitored from the headquarters building of North Cascades National Park in Sedro Woolley, Washington (C.R. Wasem, Natl. Park Serv., pers. comm.). Servheen and English (1979) and Hunt et al. (1992b) previously described eagle use of the Skagit Valley, the vicinity in which eagle 52 spent part of the winter. The westward migration of three GNP juveniles was

similar to the pattern described by Harmata and Oakleaf (1992). They reported that most of the 21 juveniles tracked from the GYE migrated westward, primarily to Washington and Oregon.

It is curious that both 1988 Lake McDonald juveniles moved independently westward from their natal area, whereas the 1991 juvenile moved north, then south. Harmata and Oakleaf (1992) and Hunt et al. (1992a) discussed the theory of genetic memory suggesting that juveniles may have a genetically-based propensity to migrate to a particular destination or in a particular direction. The 1988 and 1991 Lake McDonald juveniles were offspring of different male parents (fate of the first male was unknown). This may imply that the genetic code for migration direction is distinct in different parents, perhaps additionally suggesting disparate natal areas of the 1988 and 1991 male adults.

GNP juveniles moved west ($\bar{x} = 98$ km/d, $N = 16$ d, two eagles) and south ($\bar{x} = 87$ km/d, $N = 35$ d, five eagles) more slowly than northward migrating juveniles tracked from California ($\bar{x} = 130$ km/d, $N = 14$ d, coastal route; and $\bar{x} = 184$ km/d, $N = 21$ d, mountain route) by Hunt et al. (1992a). The five California eagles reportedly moved a considerable distance each day, whereas the GNP juveniles commonly stopped for one to several days at foraging sites. Eagle 93, from Lake McDonald, initially moved north (Fig. 3) on a route similar to that used by most spring migrants studied by McClelland et al. (1994). After traveling 408 km north in 5 d ($\bar{x} = 82$ km/d), it reversed direction, returned to GNP ($\bar{x} = 74$ km/d) for 10 d, then moved more typically southward, into Idaho ($\bar{x} = 106$ km/d in 6 d).

Lack of food in natal areas was not the major inducement for migration of GNP juveniles. Many juveniles migrating southward from Canada wintered in northwestern Montana, in some of the same vicinities that GNP juveniles vacated earlier in the autumn (McClelland et al. 1994). Additionally, the juveniles we tracked appeared to be migrating alone, apparently guided by instinct rather than following older eagles. Their parents remained on the nesting territories well beyond the juveniles' departure (Yates 1989). These factors also support the hypothesis of genetically determined migration patterns for most GNP juveniles.

Participation of GNP Juveniles at Autumn Concentrations in GNP. During the first year of our study (1985), migrating bald eagles from Canada

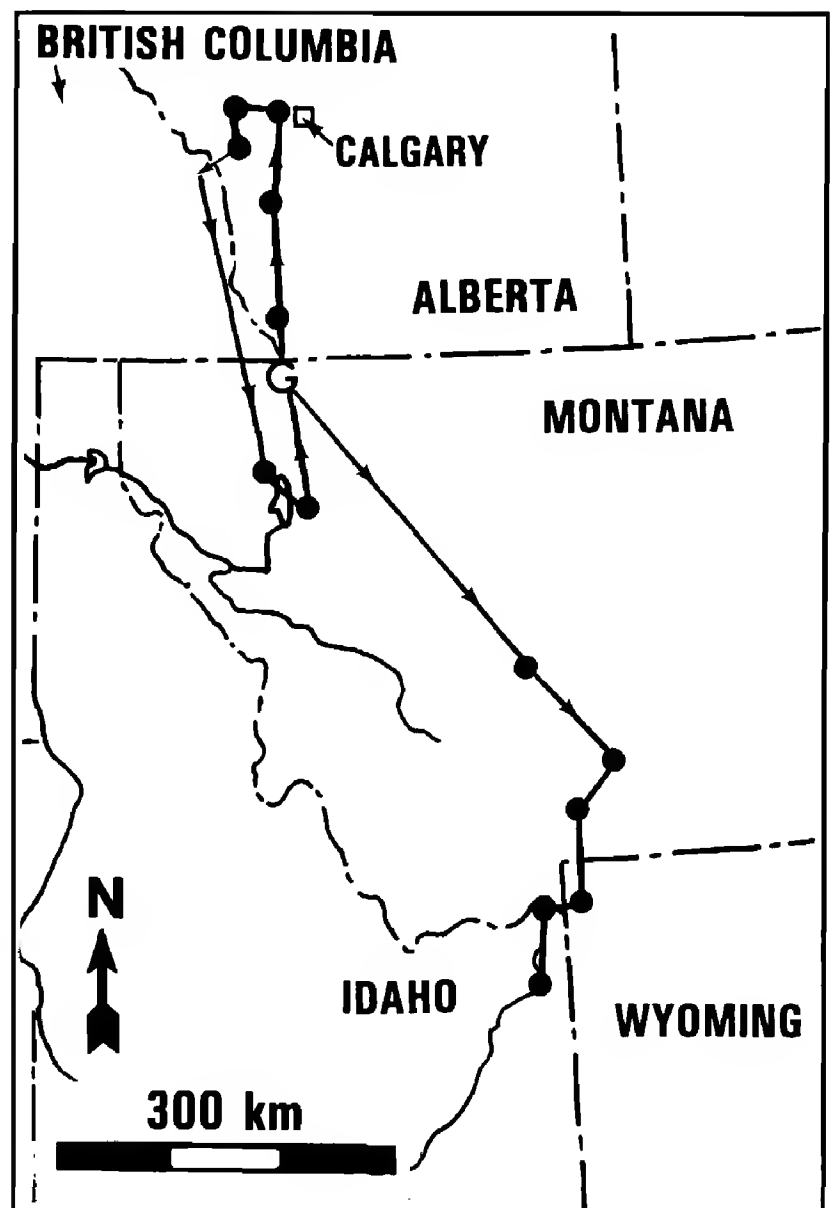


Figure 3. Autumn migration of juvenile eagle 93 in 1991, from the natal area at Lake McDonald, Glacier National Park (G symbol), Montana. After first moving 18 km east to Granite Park, GNP, eagle 93 traveled 408 km north, to the vicinity of Calgary, Alberta, Canada, then returned to Lake McDonald before southward migration to Idaho.

continued to congregate at the autumn spawning run of nonnative kokanee salmon along LMC (McClelland et al. 1982, 1994). In 1985, there were an estimated 118,000 salmon in LMC on 29 October. Eagle 01, from the Hungry Horse nest only 20 km south of the salmon run, began migration on 26 September heading south away from the concentration when there were at least 10,000 salmon in LMC. However, in the early period of spawning, few salmon were dead or easily accessible to juvenile eagles (Bennetts and McClelland 1991). The peak eagle count (520) occurred on 5 November, when eagle 01 had moved at least 360 km south of the concentration (McClelland 1992).

In 1986, the salmon population in Flathead Lake

(from which the LMC spawning run came) began a precipitous collapse (Spencer et al. 1991). However, there were still 21,500 salmon in LMC at peak count on 7 October. Two juveniles from Logging Lake, 25 km north of LMC, were potential participants at the concentration. Eagle 08 departed on 6 September. Although its migration route was unknown, we monitored for its signal daily in GNP and it did not pass near LMC. Eagle 07 flew directly over LMC on 20 September, when there were >1,000 salmon in LMC, but it continued south without stopping. Eagle 02 left Hungry Horse Reservoir on 5 October when the salmon were nearing peak numbers in LMC, but it migrated southward away from LMC. Based on the movements of these four juveniles, there seemed to be no inherent attraction to the salmon in LMC. By 1988, when we tracked the departure of five juveniles (none of which went to LMC), the salmon population had collapsed and only 120 salmon were counted in LMC; in 1991, no salmon were recorded.

Only eagle 14, in 1989, exhibited movements that might be interpreted as searching the LMC area for salmon (Fig. 2). During 17–23 October 1989, eagle 05 (from the 1988 Logging Lake nest) joined other migrating eagles at a kokanee salmon spawning run at Kikomun Creek, near Elko, British Columbia, 100 km northwest of GNP.

We concluded that most GNP juveniles did not participate in the autumn bald eagle concentrations at LMC, even during the years of salmon abundance (1963–85). Migration from natal areas was initiated before the time most salmon were available at LMC. Most GNP juveniles seem to be programmed to migrate to wintering areas far from their natal areas. Juveniles migrating southward from Canada opportunistically participated in the GNP concentration (McClelland et al. 1994), which lasted only 50 yr; perhaps this was an insufficient duration to influence inherited migration patterns of GNP juveniles.

Survival, Spring Migration, Summering Areas. Ten of the 11 radio-tagged juveniles were known to survive at least through their first winter. The fate of one juvenile was unknown. Harmata and Oakleaf (1992) reported 80% first yr survival of juveniles in the GYE. We documented spring and/or summer locations for nine juveniles (Table 3, Fig. 4). Four juveniles passed through or within several km of GNP en route to Canada. On 6 April 1990, eagles 05 and 14, moving north from different wintering sites, were both at Kintla Lake, GNP.

All juveniles moved into Canada or were moving north near the border when last located; they probably spent their first summer in Canada. Some could have returned to the U.S. after last documented locations, but we consistently monitored in GNP without success. The timing and routes of GNP juvenile migrations in spring were similar to juveniles, probably of Canadian origin, previously tracked from GNP autumn concentrations (McClelland et al. 1994). We had insufficient data to characterize the general rate of spring migration. However, eagle 06 traveled 826 km in 9 d, moving north into central Alberta from northeastern Montana. This rate ($\bar{x} = 92$ km/d) is similar to the autumn movements we documented.

During summer 1989, eagle 52 (which had wintered in western Washington) stayed at the Creston Valley Wildlife Management Area, British Columbia, Canada. On many days, its signal was monitored from the visitor center or sightings were made providing special educational opportunities for Management Area visitors (D. Ransome, Area Interpreter, pers. comm.). Through summer 1995, no marked GNP juvenile had returned and remained in a natal area. Only eagle 14 was reported in adult plumage in GNP; it was observed at Saint Mary Lake on 26 April 1995 (G. Dicus, Natl. Park Serv., pers. comm.). It was not associated with a nest and was 37 km from the Bowman Lake nest, from which it fledged in 1989. Eagle 14 apparently did not remain for the summer. Harmata and Oakleaf (1992) reported that GYE juveniles that had wintered near the west coast returned in the spring to the GYE and remained through the following autumn. Mabie et al. (1994) reported that migratory juveniles from their study area in Texas exhibited fidelity to natal nesting areas for breeding.

Foraging During Migration. Although juveniles usually were observed alone during their migration flights, they often joined other eagles at foraging sites, especially where food was concentrated. Following radio-tagged eagles allowed us to observe previously undocumented food concentrations that attracted eagles. For example, during spring migration 1989, eagle 06 was observed along the Missouri River between Ulm and Cascade, Montana, with as many as 45 other bald eagles. They foraged on ground squirrels, fish, waterfowl, and carrion (Caton et al. 1989).

We did not quantitatively assess foraging, but during migration we documented eagles feeding on or perched near carrion of antelope (*Antilocapra*).

Table 3. Telemetry and sighting locations of eight radio-tagged juvenile bald eagles from nests in and near Glacier National Park, Montana during spring and summer. Distances in parentheses are for more than one day. Last known distances from nests in bold. Eagle 14 locations (all near GNP) are not shown.

EAGLE NUMBER AND DATE	LOCATION (NORTH LATITUDE WEST LONGITUDE)	km MOVED
Eagle 01 (from the 1985 Hungry Horse nest)		
24 Apr 1986	15 km SSW Cardston, AB, Canada (49°05' 113°23')	100
Eagle 08 (from the 1986 Logging Lake nest)		
5 May 1988 ^b	20 km NW Red Deer, AB, Canada (52°23' 114°00')	—
27 May–6 Jun ^b	21 km E Edmonton, AB, Canada (53°25' 113°09')	(113)–528
Eagle 06 (from the 1988 Logging Lake nest)		
18 Mar–1 Apr 1989 ^a	Missouri River, near Cascade, MT (47°22' 111°33')	—
2 Apr ^a	Harwood Lake, 21 km SE Fort Benton, MT (47°45' 110°25')	97
3–5 Apr	Missouri River, near White Cliffs, MT (47°57' 110°05')	42
6 Apr ^a	Bearspaw Mountains, 20 km S Havre, MT (48°17' 109°38')	73
7 Apr	Soaring N at Milk River, Canada/U.S. border (49°00' 110°35')	106
12 Apr	Beaverhill Lake, 65 km E Edmonton, AB, Canada (53°24' 112°25')	(508)
13 Apr	40 km N Beaverhill Lake, AB, Canada (53°55' 112°10')	60
12–13 Jun ^a	Near Calling Lake, 89 km E Slave Lake, AB, Canada (55°20' 113°22')	(177)–762
Eagle 09 (from 1988 Cyclone Lake nest)		
13 Jun 1989	Columbia Lake, 8 km N Canal Flats, BC, Canada (50°14' 115°51')	200
Eagle 95 (from 1988 Lake McDonald nest)		
29 Apr 1989	Soaring N up Elk River, 5 km S Fernie, BC, Canada (49°28' 115°04')	129
Eagle 52 (from 1988 Lake McDonald nest)		
29 Apr–29 Aug 1989 ^a	Creston Valley Wildlife Management Area, BC, Canada (49°10' 116°35')	210
Eagle 52 (from 1988 Lake McDonald nest)		
7 Mar 1990 ^b	Pearrygin Lake, near Winthrop, WA (48°29' 120°08')	466
Eagle 05 (from the 1988 Logging Lake nest)		
28 Feb 1990 ^a	Swan River, 8 km S Swan Lake, MT (47°51' 113°50') ^c	—
12–13 Mar	Park Creek, Glacier National Park, MT (48°18' 113°36') ^c	(105)
17–19 Mar	Soaring at Salmon Prairie, Swan Valley, MT (47°37' 113°46')	(77)
6 Apr	Kintla Lake, Glacier National Park, MT (48°57' 114°08')	(155)–26
Eagle 93 (from the 1991 Lake McDonald nest)		
15 Apr 1992	First located soaring E, 21 km W Polebridge, MT (48°48' 114°33')	—
16–17 Apr ^a	Tepee Lake, near North Fork Flathead River, MT (48°55' 114°24')	18
21 Apr	Soaring NW up Cabin Creek, BC, Canada (49°10' 114°36')	(52)
24–27 Apr ^a	Columbia River, 5 km N Harrogate, BC, Canada (51°00' 116°30') ^c	(255)
28–29 Apr ^a	Columbia River, 7 km S Spillimacheen, BC, Canada (50°51' 116°21') ^c	58
30 Apr ^a	Bobbie Burns Creek, 20 km W Harrogate, BC, Canada (50°58' 116°43') ^c	31
1 May	Spillimacheen River, 8 km WSW Harrogate, BC, Canada (50°57' 116°34')	11–324

^a Sighting and telemetry location on this date.
^b Sighting only.
^c Roost location.

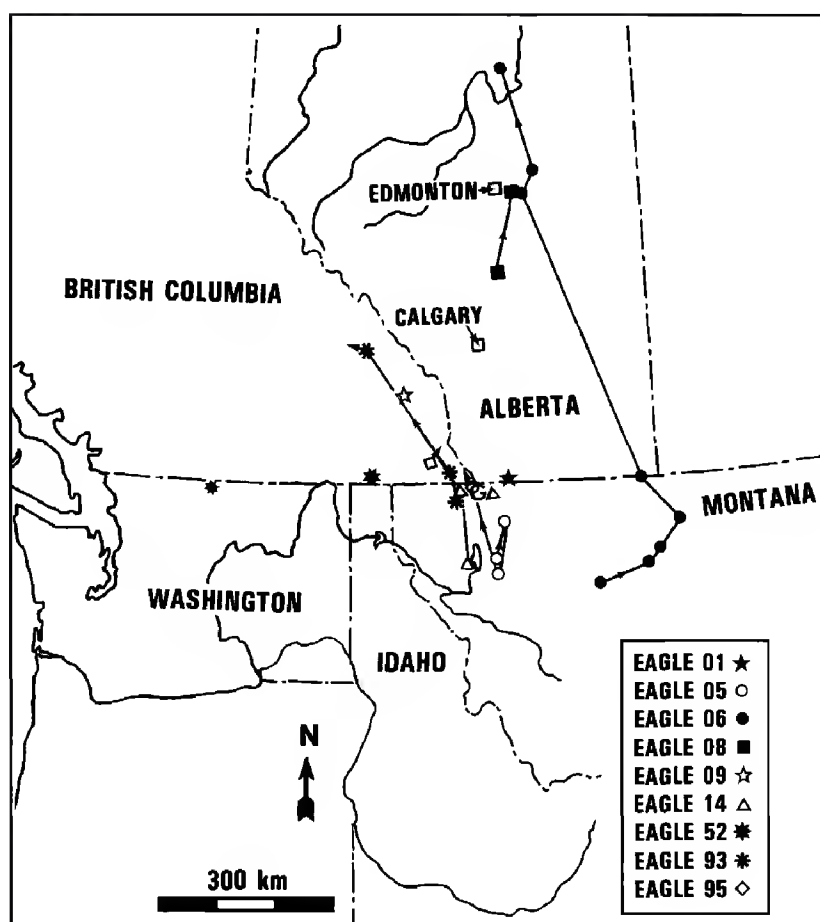


Figure 4. Spring migration routes and last known spring (or summer) locations for bald eagles from natal areas in Glacier National Park (G symbol), Montana. Large symbol for eagle 52 is 1989 summering site; small symbol is 1990 spring sighting.

pra americana), white-tailed deer (*Odocoileus virginianus*), mule deer (*Odocoileus hemionus*) and elk (*Cervus elaphus*); live and dead Richardson's ground squirrels (*Spermophilus richardsonii*); domestic cattle at a carcass dump; offal at a game farm; kokanee salmon; mountain whitefish (*Prosopium coulteri*); American coots (*Fulica americana*); and waterfowl.

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INTRA-YEAR REUSE OF GREAT HORNED OWL NEST SITES BY BARN OWLS IN EAST-CENTRAL COLORADO

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ABSTRACT.—Barn owls (*Tyto alba*) sequentially reused nest sites of great horned owls (*Bubo virginianus*) within the same breeding season on two occasions in east-central Colorado during 1982 and 1983. Two of 22 cliff nest sites used by great horned owls during the 2-year period were subsequently reused by barn owls, while no red-tailed hawk (*Buteo jamaicensis*, $N = 8$) or common raven (*Corvus corax*, $N = 20$) cliff nests were sequentially used by barn owls. In temperate latitudes, only rarely are sympatric raptor species expected to exhibit breeding behavior that could accommodate intra-year sequential nesting at the same site.

KEY WORDS: *breeding; Bubo virginianus; Colorado; barn owl; great horned owl; nest site; Tyto alba.*

Re-uso intra-anual de nidos de *Bubo virginianus* por *Tyto alba* en el centro-este de Colorado

RESUMEN.—*Tyto alba* re-usó secuencialmente nidos de *Bubo virginianus* en la misma estación reproductiva y en dos ocasiones (1982 y 1983), en el centro-este de Colorado. Dos de los 22 nidos usados por *B. virginianus*, durante el período de dos años, fueron subsecuentemente re-usados por *T. alba*. En latitudes templadas, estas especies simpátricas, raramente se espera que exhiban una conducta reproductiva que incluya nidificación secuencial intra-anual en el mismo sitio.

[Traducción de Ivan Lazo]

Barn owls (*Tyto alba*) are sympatric with great horned owls (*Bubo virginianus*) throughout much of their breeding range in North America (Johnsgard 1988), and these two species' nest-site characteristics are often similar (Knight and Smith 1982). Great horned owls are also potential predators of barn owls (e.g., Wayne 1924, Rudolph 1978, Knight and Jackman 1984, Millsap and Millsap 1987) and have reportedly killed barn owls at their nest sites (Millsap and Millsap 1987). Presumably, barn owls select nest sites that provide protection from predation by great horned owls if other factors that influence nest site selection (e.g., thermal protection, Millsap and Millsap 1987) are met.

Several authors (e.g., Smith and Marti 1976, Marti et al. 1979, Bunn et al. 1982) have suggested that barn owl populations may be limited by the availability of suitable nest sites. In predominantly

open habitats, breeding density of many raptors is thought to be limited by the availability of suitable nest sites (Olendorff and Stoddart 1974, Newton 1979, Andersen 1991). In these habitats, different raptor species often nest in closer proximity to one another than would be expected if nest sites were placed randomly or regularly (Schmutz et al. 1980, Restani 1991), and this pattern appears to correspond to the distribution and density of suitable nest sites. Where nest sites are not limiting, interspecific nest dispersion may become more regular (Rothfels and Lein 1983). Perhaps due to low availability of suitable nest sites, many raptors appropriate nests from other species from one year to the next (e.g., Newton 1979, Smith and Murphy 1982), suggesting the occurrence of interspecific competition for nest sites. However, intra-year use of the same nest site by two raptor species is probably rare.

During a 2-yr study of raptors in east-central Colorado I observed two instances of within-year use of great horned owl nest sites by barn owls. These observations are consistent with the suggestion that nest site availability may influence distribution

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Table 1. Cliff nests of large birds located on the Fort Carson Military Reservation, Colorado from 1982 through 1983 and the occurrence of intra-year reuse of those nests by barn owls.

SPECIES	YEAR	NO. OF NESTS	NO. OF NESTING AT-TEMPTS THAT FAILED		NO. OF NEST SITES REUSED
Red-tailed Hawk	1982	4	1	0	
	1983	4	0	0	
Great Horned Owl	1982	9	0	0	
	1983	13	3	2	
Common Raven	1982	10	3	0	
	1983	10	—	—	

of raptors nesting in predominantly open habitats, and provide an example of the conditions under which interspecific, intra-year breeding in the same nest site might occur. To my knowledge, similar observations have not previously been reported in the literature.

METHODS

During 1982 and 1983, I monitored nesting raptors on the Fort Carson Military Reservation (FCMR) in east-central Colorado (see Andersen et al. [1985] for a description of the FCMR). Nests were located each spring from the ground (on foot or from a vehicle) by searching potential nesting habitat. In March and/or April of each year, potential nesting areas (primarily cliff lines, prominent trees and canyons) were also surveyed from a helicopter, and all nest sites occupied in 1982 were rechecked in 1983. Each nesting attempt was monitored approximately weekly until young fledged or the nesting attempt failed. Follow-up visits were made to nests where no young fledged in order to determine cause of failure, and nest sites were visited during August in the year they were monitored to quantify their physical characteristics.

RESULTS AND DISCUSSION

During follow-up visits to nests in 1983, I located two barn owl nests at sites where great horned owls had nested earlier in the year (Table 1). One nesting attempt, located on 6 August, contained three barn owl young, and was situated where a great horned owl nesting attempt had failed on approximately 13 April 1983. The other nest contained five barn owl young and was found on 7 August at a site where great horned owls had fledged on approximately 16 May 1983. Neither barn owl nest was subsequently visited to determine nesting success, although I estimated that young were at least

halfway through the nestling period at both nests (based on photographs in Bunn et al. [1982]) and thus had a high probability of fledging.

Barn owl nesting only infrequently occurred at vacated nest sites of great horned owls (two of 22 cliff nest sites of great horned owls over a 2-yr period) and was not observed in old nests of other large cliff-nesting birds (red-tailed hawks [*Buteo jamaicensis*] or common ravens [*Corvus corax*]) on the FCMR (Table 1). Both nest sites used by barn owls were on cliffs; one in an old stick nest that appeared to have been constructed by common ravens and the other in a large natural cavity in a sandstone cliff. The only other barn owl nesting attempt located on the FCMR during the 2-yr study was in a natural cavity in a cliff (Andersen 1988).

On the FCMR, great horned owls laid eggs in early March and young fledged in early to mid-May. In north-central Colorado, barn owls fledged young from July to early September (Millsap and Millsap 1987); egg laying occurred from April through early July (Pickwell 1948, Smith et al. 1974, Colvin 1985, Marshall et al. 1986). The period of nest initiation on the FCMR is not known, but if similar to north-central Colorado, then barn owls may be able to use both successful and unsuccessful great horned owl nest sites as potential nests.

Within-year use of the same nest site by two raptor species is likely to be rare. In temperate areas, most raptors of medium to large body size begin nesting in late winter or early spring, and generally attempt nesting only once during a single breeding season (Newton 1979, Johnsgard 1988). Sequential use of a nest by different species within a single breeding season requires one species to relinquish the site early in the season, and the other to adopt the site relatively late in the season. Reuse of nest sites may also be more likely when suitable nest sites are limited and when neither species constructs its own nest, further limiting potential nest sites. These conditions are probably only met in a few species pairs, with barn owls being one of the few species likely to initiate nesting late in the season (Stewart 1952, Henny 1969).

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A COMPARISON OF BEHAVIOR AND SUCCESS RATES OF MERLINS AND PEREGRINE FALCONS WHEN HUNTING DUNLINS IN TWO COASTAL HABITATS

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ABSTRACT.—The hunting behavior and success of raptors have been linked to prey availability, habitat conditions, and competition. In this study, I compared the behavior and hunting efficiency of wintering merlins (*Falco columbarius*) and peregrine falcons (*F. peregrinus*) that hunted dunlins (*Calidris alpina*) at coastal estuarine and beach habitats in Washington to determine whether hunting efficiency was related to differences in habitat. Flocks of up to 15,000 dunlins moved 2–15 km from estuarine sites to roost and forage at beaches during diurnal high tides. Both falcon species regularly attacked flocks of dunlins in each habitat. The success rates of hunting flights for both species (merlin: 7.8%, peregrine falcon: 12.5%) were significantly lower at beaches than previously reported for estuaries. Dunlins at the beach habitat roosted in flocks near water's edge and avoided falcons by flying out over the ocean where flocks engaged in synchronized flight in the troughs between waves. Most capture attempts by falcons occurred over water even though the likelihood of successful capture was lower there. Neither age of the hunting raptor (peregrine falcon only) nor relative density of potential kleptoparasites influenced hunting success. Several hypotheses are presented to explain differences in hunting efficiency between the two habitats.

KEY WORDS: *dunlin*; *Falco columbarius*; *Falco peregrinus*; *merlin*; *peregrine falcon*; *predation*; *Washington*; *winter*.

Comparación de conductas y tasas de éxito de *Falco columbarius* y *Falco peregrinus* en la caza de *Calidris alpina* en dos hábitats costeros

RESUMEN.—Tanto la conducta de caza como el éxito de rapaces ha sido ligada a la disponibilidad de presas, condiciones del hábitat y competencia. En este estudio, comparé la conducta y la eficiencia de caza de *Falco columbarius* y *Falco peregrinus* que capturaron *Calidris alpina* en estuarios y playas, hábitat costeros de Washington, con el fin de determinar si la eficiencia de caza estaba relacionada a diferencias en hábitat. Durante la marea alta diurna, bandadas de *C. alpina*, sobre los 15.000 individuos, se movían de dos a 15 km desde el estuario a sitios de descanso y forrajeo en playas. Las dos especies de halcones atacaron regularmente a estas bandadas (*C. alpina*) en ambos tipos de hábitats. La tasa de éxito en captura al vuelo para ambas especies (*F. columbarius* = 7.8%, *F. peregrinus* = 12.5%) fue significativamente más baja en playas que en estuarios previamente reportadas. *Calidris alpina* descansó en hábitat de playas cercanos al agua y evitaron el ataque de los halcones volando hacia el mar en sincronizadas bandadas. La mayoría de los intentos de captura por halcones ocurrió sobre el agua, donde la probabilidad de éxito era la más baja. Tanto la edad del rapaz cazador (sólo *F. peregrinus*) como la densidad de potenciales kleptoparásitos no influenciaron el éxito de captura. Presento varias hipótesis para explicar las diferencias en eficiencia de caza entre los dos hábitats.

[Traducción de Ivan Lazo]

Studies of raptor foraging ecology have identified relationships between habitat use and factors such as vegetation structure and prey abundance (Wakeley 1978a, 1978b, 1979, Baker and Brooks 1981, Bechard 1982, Toland 1987, Preston 1990). They show that raptors preferentially forage in

habitats where caloric intake is higher. These findings are consistent with the ideal free model of habitat selection (Fretwell and Lucas 1970) which holds that marginal habitats will be used by certain individuals in a population to meet some or all life requisites. Few studies have attempted to describe

foraging efficiency and examine its relationship to differing habitat conditions, prey populations, or prey behavior (Swenson 1979, Bildstein 1987, Toland 1987).

In western Washington, the merlin (*Falco columbarius*) and peregrine falcon (*F. peregrinus*) are major predators of the dunlin (*Calidris alpina*; Buchanan et al. 1986, 1988, Dobler and Spencer 1989), the most abundant winter shorebird in the region (Brennan et al. 1985, Evenson and Buchanan 1995). The behavior of hunting falcons has been described for flights directed at roosting or foraging flocks of dunlins in estuaries (Buchanan et al. 1986, 1988). Large populations of wintering dunlins occur at two coastal estuaries (Buchanan and Evenson, unpubl. data); many of these birds make short flights each day during high tide to outer beach roost sites (Buchanan 1992). Individual falcons track these movements and hunt shorebirds in both habitats. To determine whether hunting efficiency was related to habitat conditions, I observed the behavior and efficiency of merlins and peregrine falcons hunting dunlins at outer beaches to compare with earlier data collected at estuaries in western Washington (Buchanan et al. 1986, 1988).

STUDY AREA

Observations were made at three beaches on the outer coast of Washington state: Copalis-North Beach (25 km in length), South Beach (23 km), and Long Beach (37 km). The beaches are contiguous, relatively flat expanses of sand backed by low dunes. Beach width during high tide ranges from 0–30 m, depending on location and tide height. The high-tide zone is characterized by open sand, varying amounts of logs, and tidal debris. Most of the length of these beaches occurs along two peninsulas situated perpendicular to the openings of Grays Harbor and Willapa Bay; only a 13 km length of South Beach is not directly across the peninsula from an intertidal area. The peninsulas are 1.5–3 km wide and are dominated on the windward side by low dunes. Dune vegetation is predominantly European beach grass (*Ammophila arenaria*) and wax myrtle (*Myrica californica*). In some areas forests of Sitka spruce (*Picea sitchensis*) and lodgepole pine (*Pinus contorta*) occur within 150 m of the dunes.

Observations of falcon hunting behavior at estuaries (Buchanan et al. 1986, 1988) were made primarily at four sites: Samish Bay in north Puget Sound, Nisqually River delta and Totten Inlet in south Puget Sound, and Grays Harbor. These sites are described in Brennan et al. (1984, 1985). Observations at beach and estuarine sites were made under a variety of weather conditions typical of the mild, wet winters in the region.

Winter populations of dunlins at three of the estuaries ranged in size from 2000–13,000 birds (Brennan et al. 1985). Counts at Grays Harbor and Willapa Bay ranged

as high as 40,000–70,000 birds, respectively (Paulson 1993, Evenson and Buchanan, unpubl. data). Although some dunlins remained within these estuaries at high tide, quite large numbers moved to the outer beaches, densities of roosting dunlins averaged >400 birds/km on beaches (Buchanan 1992). Roosting and in-flight flocks >2000 dunlins were observed in both habitats (Brennan et al. 1985, Buchanan 1992).

METHODS

Data Collection. Field work was conducted on 38 days between Nov–Feb 1983–90 and Nov–Mar 1993–94. Observations at estuaries were made in 1979–90 (Buchanan et al. 1986, 1988, unpubl. data). I traveled a cumulative total of 894 km of shoreline conducting shorebird counts and observing falcons. Each beach was visited at least once each winter with the exception of 1993–94, when South Beach was not visited. Field work was restricted to a period 3 hr before and after high tide; high tides during field visits ranged between 2.3 and 3.2 m. High tides >2.3 m inundated all tidal flats in the adjacent estuaries under most conditions. The majority of diurnal high tides during Nov–Feb were >2.3 m and ranged as high as 3.2 m.

I made most observations from a vehicle, using 7×35 binoculars and a 25× spotting scope. Certain sections of beach were covered by foot. When falcons hunted shorebirds I recorded movement and position of the flock during attack, type of hunting flight by the falcon, and the number and type of capture attempts made.

Definitions and descriptions of falcon hunting behavior are provided by Buchanan et al. (1988), and are described here briefly. A hunting flight is a perch-to-perch flight that includes one or more capture attempts. A capture attempt is defined as an attempt to seize or knock down prey. Exploratory “feints” are not considered capture attempts. Hunting methods used by merlins and peregrine falcons included stoops, nearly vertical, rapid descents toward flocks or single birds; stealth approach, rapid, low (usually <2 m) flights toward roosting flocks; and horizontal chases, the pursuit of either flocks or single birds, often after another hunting method failed.

Data Analysis. To allow comparison with other studies, I calculated the proportion of hunting flights and capture attempts that were successful. In certain cases, I observed a sequence of hunting flights involving a single falcon. Although sequential observations are not statistically independent events (Beal and Khamis 1990), they can be used to increase sample size (Hejl et al. 1990), particularly when samples are difficult to obtain, or to minimize sampling error associated with single sampling (Morrison 1984). Recently, Leger and Didrichsons (1994) found that use of pooled observations did not cause bias if intra-subject variance exceeded between-subject variance or subject sample sizes were the same.

I evaluated the dataset to determine whether inclusion of all sequential observations in the analysis was appropriate. First, I randomly selected three hunting flights per falcon and recorded the number of capture attempts per flight and calculated both intra-subject variance and between-subject variance (Leger and Didrichsons 1994). The calculated ratio of intra-subject variance to between-subject variance was 1:0.95 (i.e., intra-subject variance ex-

ceeded between-subject variance). I next calculated the number of different merlins and peregrine falcons seen hunting in each habitat. For this calculation I assumed that birds tallied between years were different birds. The numbers of falcons observed hunting in each habitat were nearly identical. At least 17 and 23 merlins hunted shorebirds at beaches and estuaries, respectively, and 11 different peregrine falcons were observed hunting in each habitat. Finally, observations at beach sites were spatially and temporally independent from those made at estuaries. For these reasons, I felt justified to use all sequential observations.

To determine whether falcons hunted with similar relative frequencies in the two habitats, I compared mean indices of hunting activity (number of flights/hr/yr) at each habitat using a Mann-Whitney test. I combined both species for this analysis because the only two estuarine sites that supported peregrine falcons were not visited all years (Buchanan et al. 1986). To test the null hypothesis that there was no difference in the efficiency of falcons hunting dunlins in different habitats, I assessed hunting efficiency in three ways. First, I compared success rates by hunting flight and by individual capture attempt for merlins and peregrine falcons at beaches and estuaries with a 2×2 chi-square contingency analysis (Zar 1984), with a correction for continuity ($df = 1$). Second, I compared the number of capture attempts per hunting flight in each habitat using the Mann-Whitney test. Finally, I compared the number of capture attempts made per flight over land and over water for both species using the Wilcoxon test.

I assessed the effects of two factors, age of falcons and presence of kleptoparasites, on hunting efficiency. Age is known to influence hunting efficiency in birds and younger, inexperienced birds generally are less successful at securing prey than adults (Burger 1988). My sample size was too small to fully evaluate the relationship between age and hunting success. However, for the peregrine falcon I used chi-square analysis to compare the proportions of hunting flights involving subadults in each habitat to determine whether differences in success rate might be explained simply by the age ratio of the birds I observed hunting in the two habitats. This analysis was not conducted for merlins because it was not always possible to determine age of hunting merlins. Because previous studies found that merlin hunting behavior varied significantly in the presence of kleptoparasites (Buchanan 1988), I attempted to determine whether the presence of known potential kleptoparasites (Brockmann and Barnard 1979) influenced the success rates of hunting flights in the two habitats. For this analysis I compared mean density indices (birds/hr/yr) of potential kleptoparasites at beach and estuarine sites using a one-tailed t -test.

RESULTS

Behavior of Dunlins at Beaches. Dunlins responded to attacks from falcons by flying directly over the ocean where flocks engaged in rapid, highly synchronized evasive movements. All predator evasion flights occurred over the water. Flocks nearly always flew very low over the water, and as-

cended and descended in an undulating motion with the passing of each cresting wave. All evasive flights at beaches occurred within 20 m of the water, and in most cases the bottom of the flock was < 1 m above water.

Behavior of Falcons. All 75 hunting flights by falcons at beaches were directed at dunlins. Hunting flights were directed at single birds and flocks ranging in size up to 12,000 dunlins. All three captures made by peregrine falcons were brief (1–2 capture attempts/flight) pursuits of solitary dunlins that had split (or were forced) away from flocks; captures made by merlins were of birds taken directly from in-flight flocks of about 4000 and 60 dunlins. Both species occasionally continued to chase birds they isolated from a flock, but no such extended pursuits were successful. Two of seven prey captures occurred over water.

Falcons were equally likely to be observed hunting in either habitat. The index of hunting incidence (hunts/hr/yr) was the same at beach ($\bar{x} = 0.64$, $SD = 0.89$) and estuarine sites ($\bar{x} = 0.55$, $SD = 0.44$; $U = 46$, $P > 0.20$).

I observed hunting flights by merlins (51) and peregrine falcons (24) directed at flocks of dunlins at beach habitat and found the success rates for hunting flights at beach sites were significantly lower than at estuaries for both species (merlin: 8 vs. 23%, $\chi^2_c = 4.17$, $P = 0.043$; peregrine falcon: 13 vs. 47%, $\chi^2_c = 4.00$, $P = 0.046$). The success rates for capture attempts were also lower at beach sites, but the differences were not significantly different (merlin: $N = 176$ capture attempts, 2 vs. 6%, $\chi^2_c = 2.42$, $P = 0.13$; peregrine falcon: $N = 52$ capture attempts, 6 vs. 15%, $\chi^2_c = 2.26$, $P = 0.15$).

The number of capture attempts per flight at beach sites (merlin: $\bar{x} = 3.45$, $SD = 3.52$; peregrine falcon: $\bar{x} = 3.47$, $SD = 2.92$) were the same as at estuarine sites (merlin: $\bar{x} = 3.81$, $SD = 4.77$, $Z = 0.04$, $P > 0.50$; peregrine falcon: $\bar{x} = 2.75$, $SD = 2.4$; $U = 152.5$, $P > 0.20$). Both falcons made fewer capture attempts per flight over land (merlin: $\bar{x} = 0.69$, $SD = 0.68$; peregrine falcon: $\bar{x} = 0.5$, $SD = 1.06$) than over water (merlin: $\bar{x} = 2.78$, $SD = 3.6$, $T_- = 248$, $P < 0.001$; peregrine falcon: $\bar{x} = 2.5$, $SD = 2.25$, $T_- = 3$, $P < 0.001$). All four prey captures by merlins occurred over the beach. Two of three captures of prey by peregrine falcons occurred over the water when single birds were captured away from flocks; the third capture occurred when a single bird was taken above the beach after leaving a flock out over the water. The success rates of

capture attempts made over beaches by merlins and peregrine falcons were 11.4 and 8.3%, respectively. Only 5% of capture attempts by peregrine falcons over water were successful.

When attacking a flock over the beach, both species always used low stealth approaches above land; there were no initial approaches over water. In some cases, the flock saw an approaching falcon ≈ 250 m away and moved to a position over the breakers where much of the hunting activity occurred. In such cases, the falcon either continued the attack directly or circled to a position above the breakers (merlins: 20–50 m, peregrine falcons: 40–80 m), from where they initiated stoops or discontinued the attack. For merlins, 17 flights occurred exclusively over the beach, 12 were initiated over the beach and moved to water (e.g., capture attempts were made above beach and water during a hunting flight), and 22 occurred exclusively over water (e.g., the capture attempt occurred over water). This pattern of hunting location differed slightly for peregrine falcons, where three flights occurred exclusively over beach, four were initiated over beach and moved to water, and 17 occurred exclusively over water (2×3 contingency test; $\chi^2 = 5.42$, $P = 0.07$). Low attacks occurred in 38 (75%) and 10 (42%) of the hunting flights by merlins and peregrine falcons, respectively; most low approaches were stealth flights, but some flights over water (merlin = 10, peregrine falcon = 5) also included low pursuit of single birds. All stoops by both species were made over water and occurred in most flights that were exclusively above water (merlin: 16 of 22, 73%, peregrine falcon: 15 of 17, 88%).

Influences on Hunting Efficiency. Nine of 24 hunting flights by peregrine falcons at beaches (38%) were made by subadult birds. This is similar to the proportion of flights by subadults observed at estuaries (33%; $\chi^2_c = 0.08$, $P = 0.78$).

I observed six raptor species in the two habitats that are known to steal prey from other raptors: bald eagle (*Haliaeetus leucocephalus*), northern harrier (*Circus cyaneus*), red-tailed hawk (*Buteo jamaicensis*), rough-legged hawk (*B. lagopus*), gyrfalcon (*F. rusticolus*), and peregrine falcon. As a group, these species were observed at a much lower frequency (birds/hr/yr) at beaches ($\bar{x} = 0.34$, $SD = 0.22$) than at estuaries ($\bar{x} = 1.42$, $SD = 0.4$; $t = 6.6$, $df = 1, 14$, $P < 0.0005$). I observed only one instance of attempted kleptoparasitism at beach sites

(a gyrfalcon attempted to take prey from a peregrine falcon immediately after capture).

DISCUSSION

Merlins and peregrine falcons hunted less efficiently, in terms of the success rate of hunting flights, at beaches (8 and 13%, respectively) than at estuarine sites (23 and 47%, respectively). Both species had similar success rates for capture attempts at beach sites (2 and 6%, respectively) and estuaries (6 and 15%, respectively) but peregrine falcons were more successful than merlins at capturing dunlins. Although peregrine falcons are often far more successful at securing prey (Ratcliffe 1980), relatively low rates of success have been documented (Bertochi et al. 1984, see review by Roalkvam 1985). The variation in rates of successful hunting flights by peregrine falcons and merlins have been attributed to factors such as age, experience, degree of intent, and energy requirements (Ratcliffe 1980, Treleven 1980, Sodhi et al. 1993); factors associated with the hunting success rates of other raptor species include the behavior and vulnerability of prey and vegetation structure (Swenson 1979, Bildstein 1987, Toland 1987).

Solitary shorebirds can be more vulnerable to predation than birds in flocks (Kus 1985). Species reliant upon synchronized flocking when threatened by predators perhaps benefit from this behavior by reducing the probability of predation (Kus 1985), by confusing the predator (Davis 1980). In this study I found that five of seven prey captures were of single birds isolated from flocks. Most captures occurred above beaches with only two captures occurring over water. There were no successful captures made through attacks on large flocks over water. This was in striking contrast to hunting behavior at estuaries where 43 and 32% of all prey captures by peregrine falcons and merlins, respectively, occurred during stoops at flocks of dunlins (Buchanan et al. 1986, 1988). In both habitats, attacks at flocks resulted in single birds becoming isolated from the flock and these were most successful when dunlins were not flying over waves.

Shorebirds are known to evade predators by flying over water (Hunt et al. 1975, Bertochi et al. 1984, Boyce 1985, Buchanan et al. 1988). The height of dunlin evasive flights I observed at beaches was lower than at estuaries (e.g., 50 m; Buchanan et al. 1988), suggesting an adaptive advantage of dunlins flying low over the water to evade fal-

cons. There are several possible reasons why shorebirds evade predators by flying over water and why falcons are less effective when hunting shorebirds in beach habitat. First, shorebirds are capable of safely landing in water to escape capture (Hunt et al. 1975, Buchanan et al. 1991). Although both peregrine falcons and merlins are known to retrieve floating or swimming birds from water (Dekker 1980, Boyce 1985, Buchanan et al. 1991), they may not be able to do so in turbulent water (e.g., the breaker zone). If falcons are reluctant to retrieve such birds, perhaps they must attempt more difficult captures of prey in mid-air. This is an unlikely explanation because falcons typically secure shorebirds in flight. Second, despite the fact that merlins and peregrine falcons occasionally hit the water at the terminus of stoops during hunting flights (three observations for each species at estuaries; J.B. Buchanan, unpubl. data), the risk of hitting a wave during a stoop, or of being inundated by a cresting wave may influence a falcon's ability to capture prey so close to the water. It is difficult to assess the potential importance of this factor. Third, falcons may simply be confused by the contrasting movements of individuals within the flock relative to the movement of the waves and may have difficulty tracking target birds. Predator confusion is one reason why prey species evade predators by assembling in large flocks capable of cohesive movements (Curio 1976, Davis 1980). Falcons in this study were unsuccessful in all attempts to capture dunlins during stoops over cresting waves even though stoops over calm water were commonly made and often successful at estuaries (Buchanan et al. 1988). Successful captures over water occurred during horizontal pursuit of birds isolated from flocks; it is likely that any confusion effect is negligible during horizontal pursuit of a single bird. These results appear to support a predator-confusion hypothesis. Finally, the lower rate of hunting success at beaches did not appear to be related to density of potential kleptoparasites, which were more abundant at estuarine sites.

According to the ideal free model of habitat selection (Fretwell and Lucas 1970), both dunlins and falcons should attempt to spend as much time as possible in habitats that serve to maximize inclusive fitness. This means that shorebirds should forage in habitats where invertebrate prey is available at a level that offsets physiological costs associated with roosting, vigilance, and predator avoidance. Similarly, merlins and peregrine falcons

should hunt in areas where prey is abundant and available enough to offset costs associated with foraging. Falcons may hunt in beach habitat, where they are less efficient predators, in order to meet their high energetic costs. A merlin, for example, requires about 70–75 g food each day (Page and Whitacre 1975), and in western Washington would need to consume about 3 dunlins (Brennan et al. 1984) or similar-sized prey every 2 days to meet this intake requirement. Owing to the short daylength and very high diurnal tides during winter, shorebirds may spend as much or more time at beach sites than at estuaries on certain days. Therefore, a merlin may need to hunt shorebirds in less optimal conditions to meet its average daily energy requirement, unless other prey are available. This expended effort may be offset by the greater success rate for hunting flights at estuaries, and the fact that falcons can more easily track flock movement between habitats (e.g., search time is reduced). However, it is unknown whether the lower success rate of hunting flights at beaches represents a significant physiological stress to falcons and whether other potentially suitable prey are available during the periods when dunlins are roosting at outer beaches. These issues must be addressed to better understand these predator-prey relationships.

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SHORT COMMUNICATIONS

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HABITAT PREFERENCE OF CRESTED SERPENT EAGLES IN SOUTHERN JAPAN

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KEY WORDS: *conservation; crested serpent eagle; Spilornis cheela; habitat preference; wet grasslands; Japan.*

The crested serpent eagle (*Spilornis cheela*) is a medium-sized raptor whose range includes most of the Indo-oriental region (Brown and Amadon 1968). Over 20 subspecies are recognized, all of which are associated with forests of tropical and subtropical regions (Brown and Amadon 1968). The Japanese race (*S. c. perplexus*) is of uncertain taxonomic status but considered a separate species by some authors (Devillers et al. 1993). The serpent eagle's Japanese range is restricted to the subtropical islands of Ishigaki and Iriomote, where it is typically associated with the wet grasslands of the southern end of Japan's Ryukyu Island chain. Because the population contains as few as 100 eagles (Hanawa et al. 1985), it has been listed as a "Species of Concern" by the Environment Agency of Japan (Environment Agency of Japan 1991). Lack of information on the habitat requirements of serpent eagles in this rather unusual habitat has delayed the development of management plans for its conservation. Herein, we provide evidence for the winter habitat preference of this eagle on Ishigaki and Iriomote Islands.

STUDY AREA AND METHODS

Ishigaki and Iriomote Islands (24°N, 124°E) of southern Japan are subtropical islands of continental origin. Recently, Ishigaki Island has undergone widespread conversion of its natural forest and historical wet rice agriculture to sugar cane production. On Iriomote Island, less of this conversion has occurred and it continues to support intact mangrove (*Rhizophora* spp.) and tropical broad-leaved forests.

We collected data on the distribution of perched crested serpent eagles along survey routes from 10–15 February 1993. We rode motorcycles at 20–30 km/h to survey for eagles. On Iriomote, two routes (9.9 and 14 km each) were surveyed twice a day by two observers between 0700–1000 H (a total of 4 times/d) for 3 d. On Ishigaki Island, one circular route (34.5 km in length) was surveyed in the same way. We plotted all eagle locations on maps, and recorded the perch habitat, perch structure, and the distance from the nearest forest edge for each

perched eagle observed. Habitats were separated into six categories: wet grasslands (including wet grass fields and wet meadow/rice cultivation), dry grasslands (including dry, grass fields and vegetable farms), grazed pastures, forests, sugar cane fields, and residential areas.

Dominant land use within 50 m of either side of the survey routes was estimated visually during surveys and compared to published maps of land use (Environment Agency of Japan 1981).

RESULTS AND DISCUSSION

We observed 97 perched serpent eagles during surveys. The most commonly used perch structures were trees ($N = 58$) followed by utility poles ($N = 39$). Eagles typically were perched within 10 m of the forest edge, and were less frequently observed 10–50 m from the closest forest edge (Fig. 1). Average distance from forest edge was 4.3 m (SD = 10.9 m, $N = 77$) on Iriomote Island, and 11.9 m (SD = 20.3 m, $N = 20$) on Ishigaki Island, but this difference was not significant (Mann-Whitney $U = 602$, $Z = 1.73$, $N_1 = 76$, $N_2 = 20$, $P > 0.05$).

Perching habits of eagles were similar on the two islands (Fig. 2). Most eagles perched in wet grasslands on Iriomote (67.5%) and Ishigaki (80%) Islands. Based on the estimated availability of wet grasslands, serpent eagles used this habitat more than expected (Iriomote Island

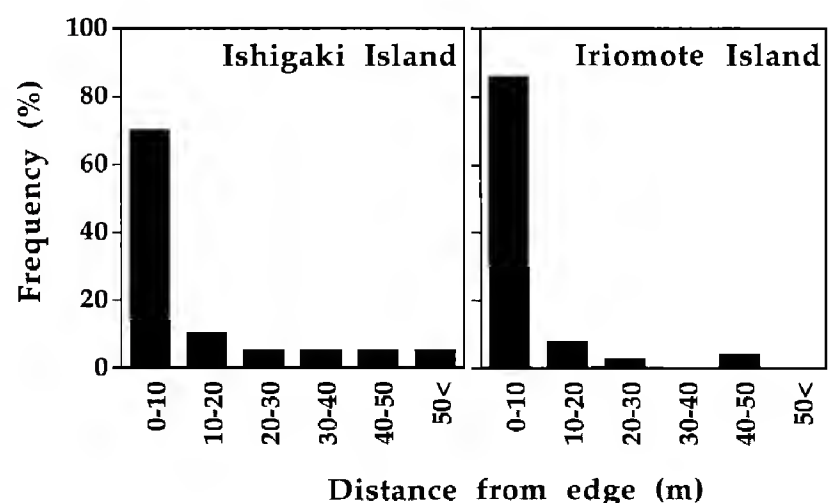


Figure 1. Perch sites of crested serpent eagles and distance categories from forest edges of Ishigaki ($N = 20$) and Iriomote Islands ($N = 77$), southern Japan.

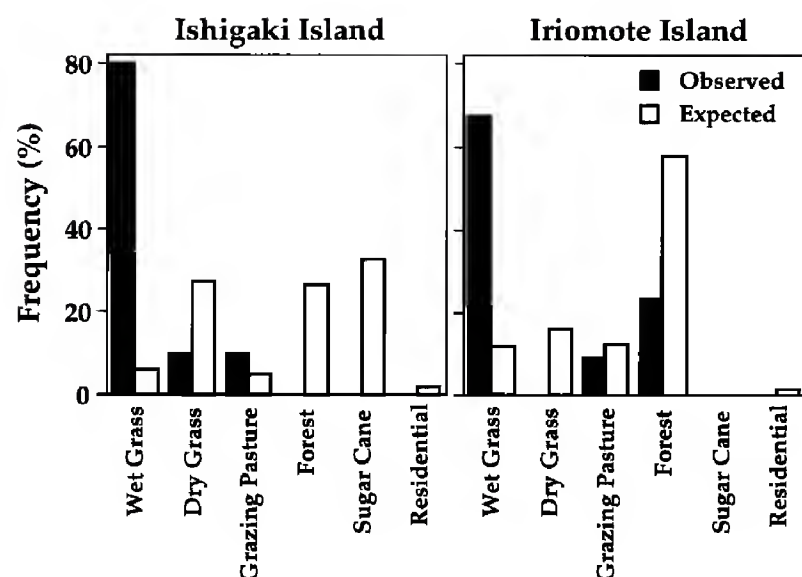


Figure 2. Habitat preference of crested serpent eagles on Ishigaki ($N = 20$) and Iriomote Islands ($N = 77$), southern Japan.

$\chi^2 = 225.4$, $P < 0.001$; Ishigaki Island: Binomial test, $P = 0.000$).

Our results showed that margins of forests and wet grasslands served as important perching habitats for crested serpent eagles wintering on the Iriomote and Ishigaki Islands of Japan. There were an estimated 10 000 ha of wet grasslands along survey routes on Iriomote Island and 12 500 ha on Ishigaki Island in 1981. In 1992, there were only 5600 and 7100 ha in the same areas of Iriomote and Ishigaki Islands, respectively. Most wet grasslands had been converted to sugarcane fields or pastures. It appeared that, if further conversion of wet grasslands occurred, it would threaten the future status of the serpent eagle in this portion of its range.

RESUMEN.—Se estudiaron las preferencias de hábitat de *Spilornis cheela* en las islas de Ishigaki e Iriomote, ubicadas

al sur de Japón. La águila prefirieron los márgenes de bosques y praderas húmedas como sitios de percha. Las praderas húmedas fueron convertidas en campos de caña de azúcar y de pastura. Esta águila está listada como una “especie de interés” por la Agencia del Medioambiente de Japón. Por lo tanto, futuras conversiones de praderas para uso agrícola deben considerar la importancia de este hábitat para *S. cheela*.

[Traducción de Ivan Lazo]

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A POSSIBLE CASE OF POLYANDRY IN MONTAGU'S HARRIER

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KEY WORDS: polyandrous association; Montagu's harrier, *Circus pygargus*; Spain.

Monogamy is the most common mating system among raptors (Newton 1979), although alternative mating systems such as polygyny have been described in several species (see Newton 1979). In contrast, polyandry is rare and has only been described in the Harris'

hawk (*Parabuteo unicinctus*) (Bednarz 1987), Galapagos hawk (*Buteo galapagoensis*) (Faaborg et al. 1980), and bearded vulture (*Gypaetus barbatus*) (Heredia and Donazar 1990). Polyandry has been described as occasionally occurring in species including kestrels (*Falco tinnunculus*) (Packham 1985), golden eagles (*Aquila chrysaetos*) (Berg 1988), pygmy falcons (*Polihierax semitorquatus*) (Thomsett 1991), and Egyptian vultures (*Neophron*

percnopterus) (Tella 1993). Polyandry is usually associated with circumstances when male breeding becomes restricted by the availability of food, mates, or nesting places (Newton 1979).

Montagu's harrier (*Circus pygargus*) is generally considered a monogamous species (Cramp and Simmons 1980), although bigyny has been recorded occasionally in the Netherlands (Hens 1926 in Cramp and Simmons 1980) and in England at very low frequencies: 1/71 nests in Cornwall (Khan in Cramp & Simmons 1980), 13/776 in Norfolk in 1923–1982 (Day 1990), 1/25 pairs in Norfolk in 1980–86 (Day 1990).

I studied an unmarked population of about 50 pairs of Montagu's harriers between 1991–1995 in northeastern Madrid (central Spain) (Arroyo 1995). It nested in cereal crop fields, and nest dispersion was clumped with pairs forming aggregations of 2–16 pairs and a median distance to the nearest neighbor being 280 m (range = 30–4300 m). As in other populations of Montagu's harriers, monogamy was the predominant mating system.

In 1992, a possible polyandrous association was observed at one nest. On 20 June, two adult males (>2 years old, as determined by plumage) were observed simultaneously bringing food to 12–16-day-old nestlings in a single nest. The first male arrived with prey at about 1600 H. The female left the nest to take the food, after which both birds perched near the nest. While the female was eating and the first male was still perched, a second male arrived with food. The female flew to this second male and took the food in a typical aerial food pass. When both birds entered the nest together, the first male was still perched in view. Between 21–25 June, the nest was observed for a total of 15 hr, during which 13 food deliveries were observed. This food delivery rate was higher than at other nests (\bar{x} = 0.36, SE = 0.08 prey deliveries/hr, N = 14 nests). On four occasions both males brought food to this nest at the same time. In one instance, one of the males brought the food directly to the nestlings, while the female was engaged in an aerial food pass with the other male. On three occasions, the male (or males) waited 2–15 min until the female left the nest to take the food. This suggested that either the chicks were satiated, or that the combined feeding rate of the two males was greater than what the female could provide to the chicks.

Aggression between the two males was observed only once, with one of the males skydancing (Hamerstrom 1969) in front of the other. Both males also attacked a plastic crow which was placed near the nest from 21–25 June. The first two days both males attacked the crow as soon as they saw it. On 25 June, neither of them attacked the decoy, but both screamed at it until it was removed.

It is not known whether provisioning by both males was also made at the incubation stage, as the nest was observed for only two days in that period. On 28 May, the nest was observed between 0800 and 1000; one male was observed bringing food at 0830, but the female did not come out of the nest to take the food until 90 min later.

The male perched nearby with the prey during this time, flying over the nest from time to time with the prey, and occasionally eating some of the food. On 30 May, the female was fed twice within 20 min, but the male had disappeared after the first food delivery, so it was not known if both deliveries were by the same male or not.

The nest where these observations took place was 890 m from the next nearest known nest. During the pre-laying period, I observed this nest from 23–30 April and on 5 May for 1 hr each day. On the first day, two females (identifiable by plumage differences) and two males were present, and many aggressive interactions between all individuals were observed. From 30 April onward, only one pair was observed in the area, and a second nest was never found. The female was observed copulating during only two observation periods but this rate was higher than that of other pairs at a similar breeding stage.

Polyandrous associations have previously been described in Montagu's harrier in France (Cormier 1990) and England (Khan in Cramp & Simmons 1980). The case in England involved a first-summer male assisting with the provisioning of a mated female. In France, it involved a 2-yr male that had never bred before, arrived late in the season and contributed to the provisioning of a female with which it was not seen to copulate. Young males may benefit from helping already-mated females by improving their opportunity to mate with them in future years. However, remating with partners from previous years has been only rarely observed in a long-term study on wing-tagged Montagu's harriers (Bretagnolle pers. comm.), so the benefit of caring for the offspring of potential future mates may be small. In both previous records of polyandrous associations, the extra male was suggested to benefit by gaining experience for subsequent breeding attempts (Cormier 1990). The case observed in this study could have a similar explanation. However, it is also possible that both males had copulated with the female, and the second male was potentially provisioning some of his own offspring. About 4–7% of 139 copulations observed in the study area from 1992–95 were extra-pair copulations, suggesting that one of the males in the polyandrous association may have been an unmated partner that remained to help provision the chicks.

Polyandry seems to occur only occasionally in Montagu's harrier. A more detailed study with a marked population of breeding harriers would give more insight into the frequency and circumstances in which this mating system arises.

RESUMEN.—Esta nota describe una observación de un trío poliándrico en el aguilucho cenizo (*Circus pygargus*), en la que dos machos adultos alimentaron y defendieron la pollada de una misma hembra. No se tienen datos sobre el comportamiento de cópulas de estos tres individuos, pero se especula sobre la posibilidad de que ambos machos compartieran la paternidad de la pollada,

dado que se han observado cópulas extra-pareja en la población estudiada.

[Traducción Autor]

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NOTES ON THE DIET OF SHORT-EARED OWLS (*ASIO FLAMMEUS*) IN TEXAS

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KEY WORDS: Texas; short-eared owl; *Asio flammeus*; diet.

Although a common winter resident along the gulf coastal plain (Oberholser 1974, Rappole and Blacklock 1994), no information exists on the diet of short-eared owls (*Asio flammeus*) in Texas. This lack of information is in stark contrast to the plethora of dietary information for the species from other portions of its range (Tomkins 1936, Banfield 1947, Stegeman 1957, Munyer 1966, Clark 1975, Wiebe 1991, Rau et al. 1992, Holt 1993, Holt and Leasure 1993). Here we report the results of an analysis

of short-eared owl pellets collected in the Lower Rio Grande Valley, Texas.

Between 27 February and 3 March 1995, short-eared owl pellets were collected from the Marinoff Tract, Lower Rio Grande National Wildlife Refuge, Hidalgo County, Texas. As a result of activities associated with an ongoing vegetation study, roosting short-eared owls were flushed periodically allowing for the identification of roost sites and collection of pellets. Roost sites were located on the ground in grasslands dominated by dense stands of guinea grass (*Panicum maximum*) (Gould 1975) with a mean height of 60 cm. Pellets were collected daily from each roost site until abandoned by the owls.

Pellets were dissected and prey remains collected after submerging the pellets in a 1.0% (w/v) solution of sodium chloride. After approximately 10 min in the salt solution, pellets were teased apart and allowed to remain

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in solution for another 10 min. The floating material was then collected with a fine metal screen and the remaining liquid was poured through a fine net sieve. This process was repeated until all skeletal material was separated from the hair, feathers, or detritus in the pellets. The hair, feathers and skeletal material were then dried in a warm oven, and examined with the aid of a dissecting microscope.

Mammalian skull and dental remains found in pellets were identified using a skull key (Jones and Manning 1992). Avian remains were identified with the aid of comparative material in the Texas Cooperative Wildlife Collection at Texas A&M University. Insects were identified to order with the aid of an insect field guide (Borror and White 1970) and the reference collection at the Santa Ana National Wildlife Refuge. Estimates of biomass were calculated for each prey item identified to species level. Average mass estimates used to calculate biomass were derived from Lowery (1981) and Davis and Schmidly (1994).

Three roost sites were located during the course of this study. Since none of the owls flushed from these roosts were banded and/or color marked we could not determine conclusively the number of owls using each roost during the observation period. From these roost sites, a total of 48 short-eared owl pellets were collected. Based on the remains in each pellet, 38 prey items were identified. The majority of prey items, 29 (76.3%), were mammalian. Of the total prey remains found, the least shrew (*Cryptotis parva*) accounted for over half, 22 (57.9%), of the prey items observed. Additional mammal remains included three (7.9%) white-footed mice (*Peromyscus leucopus*), two (5.3%) hispid cotton rats (*Sigmodon hispidus*), one (2.6%) house mouse (*Mus musculus*), and one (2.6%) Mexican spiny mouse (*Liomys irroratus*). The remaining prey items included one (2.6%) unidentified Icteridae bird, and eight (21.1%) grasshoppers of the order Orthoptera.

In terms of biomass, hispid cotton rats contributed approximately half (44.5%) of the total followed by least shrews (25.6%), white-footed mice (15.4%), Mexican spiny mice (10.9%) and house mice (3.6%). The large contribution of cotton rats to total dietary biomass supports the inverse relationship between prey size and the total number of prey items per pellet (Weller et al. 1963) and may explain the low mean number of prey items per pellet observed in this study (0.792) relative to previous studies (Clark 1975, Wiebe 1991).

Our analysis provides two noteworthy observations regarding the diet of short-eared owls in Texas. The remains of least shrew and Mexican spiny mouse collected are the first reported occurrence of these species in the diet of short-eared owls. While our results support previous observations that mammals are the principle component in the diet of short-eared owls (Holt 1993, Holt and Leasure 1993), the large portion of shrews encountered differs from previous observations and suggests

that in areas devoid of microtines, shrews are an important food item.

The number of insects, especially grasshoppers, encountered in the pellets of these wintering short-eared owls was also of interest. Previous investigations of the food habitats of short-eared owls suggest that insects constitute a minor percentage of prey items taken by the owls (Wiebe 1991, Rau et al. 1992, Holt 1993, Holt and Leasure 1993). Because we were unable to identify the grasshopper remains to species, no attempt was made to quantify the contribution of the insects taken in terms of total dietary biomass. Our results suggest, however, that when present grasshoppers may contribute to the diet of wintering short-eared owls in south Texas.

RESUMEN.—Los restos de fauna obtenidos del análisis de 48 egagrópilas de *Asio flammeus* mostraron que la mayoría de las presas consumidas, 29 (76.3%), eran mamíferos. Del total, más de la mitad 22 (57.9%) correspondían a *Cryptotis parva*. Adicionales restos de mamíferos incluyeron tres *Peromyscus leucopus* (7.9%), dos *Sigmodon hispidus* (5.3%), un *Mus musculus* (2.6%) y un *Liomys irroratus*. Las categorías de presas restantes incluyeron un Icteridae (2.6%) y ocho saltamontes (21.8%) del orden Orthoptera. Nuestros resultados documentan la primera ocurrencia de *C. parva* y *L. irroratus* en la dieta de *A. flammeus* y sugieren que los saltamontes son más comunes en la dieta de lo que previamente ha sido reportado.

[Traducción de Ivan Lazo]

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LETTERS

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GULLS (*LARUS* spp.) IN THE DIET OF FERRUGINOUS HAWKS

Ferruginous hawks (*Buteo regalis*) eat various types of prey, but in most areas rely heavily on just a few species such as jackrabbits (*Lepus* spp.), ground squirrels (*Spermophilus* spp.), pocket gophers (*Thomomys* spp.), and prairie dogs (*Cynomys* spp.) [M. J. Bechard and J. K. Schmutz 1995, *The Birds of North America*, No. 172, A. Poole and F. Gill, eds.]. Twenty studies examining the dietary habits of ferruginous hawks have identified 6,203 prey items [R. R. Olendorff 1993, U. S. Dept. Inter., Bur. of Land Manage., Boise, ID.]. Mammals comprise 95.3% of the prey taken by biomass (83.3% by frequency) and passerines and other birds comprise only 4.1% of the overall diet by biomass (13.2% by frequency). Shorebirds are taken infrequently and account for only approximately 0.09% of the overall diet by biomass (0.08% by frequency). Gulls (*Larus* spp.) have never been reported in the diet of ferruginous hawks, despite the fact that they are a common species that occurs throughout much of the hawk's breeding range. Herein, we document the occurrence of both California (*Larus californicus*) and ring-billed (*Larus delawarensis*) gulls in the diet of ferruginous hawks in Washington state during the breeding season.

In 1994 and 1995, we collected pellets from occupied ferruginous hawk nest sites on and adjacent to the U. S. Department of Energy's Hanford Site in southcentral Washington (Benton County). While collecting pellets, we observed piles of gull remains at 3 nest sites. At one site, remains from at least eight gulls were found in 1994, and numerous piles were again found at the same nest in 1995. Piles were scattered on the ground up to 50 m from the nest. Pellets collected at this nest contained mandibles, feet, and feathers from gulls verifying that the ferruginous hawks, and not some other mammalian predator, were indeed eating both species of gulls. Remains were mainly those of juvenile birds.

At a second nest site, remains from at least 3 gulls were found scattered near the nest in 1994. Again, evidence in pellets verified that the gulls had been eaten by the ferruginous hawks. We did not return to collect pellets at this nest in 1995. At a third nest in 1995, a gull wing was seen hanging from the edge of the nest during the nesting period. After the young had fledged, we returned and found several piles of gull remains scattered near the base of the nest structure.

Gulls were common in the study area during spring and summer months and were frequently observed in large groups foraging on crickets. There were also several breeding colonies in the study area. Under conditions such as these, it appears that gulls can provide an alternative food source for ferruginous hawks that has not been previously reported.

These observations were made during a study on ferruginous hawks that was funded by the U. S. Department of Energy under contract number DE-AC06-76RLO-1830, and Associated Western Universities Northwest. We would like to thank the Washington Department of Fish and Wildlife for assistance with field work during our studies of ferruginous hawks in Washington and Marc J. Bechard for logistical support. —Alan W. Leary, Aimee L. Jerman, Raptor Research Center, Dept. of Biology, Boise State Univ., Boise, ID 83725 U.S.A., and Rosemary Mazaika, Battelle Pacific Northwest Laboratory, P.O. Box 999, Richland, WA 99352 U.S.A.

J. Raptor Res. 30(2):105–106

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STOMACH CONTENTS OF A SWAINSON'S HAWK FROM ARGENTINA

The Swainson's Hawk (*Buteo swainsoni*) is known to feed on a wide array of prey including mammals, birds, reptiles and insects (J.C. Bednarz 1988. *Condor* 90:311–323; J.K. Schmutz et al. 1980. *Can. J. Zool.* 58:1075–1089). In Argentina,

the species is known as the “aguilucho langostero” (grasshopper hawk) due to the fact that grasshoppers are considered to be an important prey item (B. Woodbridge et al. 1995. *J. Raptor Res.* 29:202–204). Observations on predation are scarce in Argentina, although an unusual incident has been reported (D.C. Rudolph 1993. *Wilson Bull.* 105:365–366).

A specimen was shot 11 January 1991 by a rural worker in Monte Nievas, Conhelo department, La Pampa, Argentina, and brought to the Museo Provincial de Historia Natural of Santa Rosa, La Pampa, where it was prepared as a study skin and the stomach preserved. An analysis of the stomach contents revealed a total of 40 prey items, of which most were grasshoppers (26 *Dichroplus elongatus*, 2 *Xileus* spp.). The remainder were either undetermined grasshoppers (2 adults), lepidopterans (8 larvae) or chrysomelid beetles (1 adult, 1 larva).

It is obvious that in this case, acridid grasshoppers (called “tucuras” in Argentina) were an important and well represented item in the diet of this bird. The genus *Dichroplus* of acridid grasshoppers has been reported in Swainson's hawks pellets and have been mentioned as a hazard for hawks because of insecticides used against them (B. Woodbridge et al. 1995. *J. Raptor Res.* 29:202–204).

The bird was a female that had been banded near Edmonton, Alberta, Canada, in July 1982. This is further evidence that this portion of La Pampa province is used as the wintering grounds by Swainson's hawks from western Canada.

We wish to thank G. B. Siegenthaler for permission to study the specimen.—**Ramón Serracín Araujo, Departamento de Ciencias Naturales, Universidad Nacional de La Pampa, Uruguay 151, 6300 Santa Rosa, La Pampa, Argentina and Sergio I. Tiranti, Department of Biological Sciences, Texas Tech University, Lubbock, TX 79409-3131 U.S.A.**

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AN ASSESSMENT OF MORTALITY OF SWAINSON'S HAWKS ON WINTERING GROUNDS IN ARGENTINA

Swainson's hawks (*Buteo swainsoni*) migrate from breeding areas in grasslands and shrubsteppe areas of North America to spend the austral summer in similar habitat in Argentina (White et al. 1989, *Raptors in the Modern World*, Berlin, Germany). The overall population, estimated at 450,000 birds, appears to be stable although declines have been reported in California (P.H. Bloom 1979, USDI BLM Dept. Fish and Game W-54-R-12, Sacramento, CA U.S.A.), Oregon (C.D. Littlefield et al. 1984, *Raptor Res.* 8:1–5), Nevada (G.B. Herron and P.B. Lucas 1978, Nev. Dept. Fish and Game, Perform. Rept. W-43-R, Reno NV U.S.A.), and Alberta and Saskatchewan (C.S. Houston and J.K. Schmutz 1995, *J. Raptor Res.* 29:198–201).

To better describe the migration route of the Swainson's hawk, two satellite radio transmitters were attached to females in 1994. Following radio-transmitter locations to La Pampa Province in Argentina, a roost with over 700 dead Swainson's hawks was encountered in 1995 (Woodbridge et al. 1995, *J. Raptor Res.* 29:202–204). The hawks arrived at the roost after feeding in a sunflower field and died during the following three days. The landowner stated that the field had been sprayed with the organophosphate (OP) monocrotophos. Although monocrotophos is used widely abroad, it is not registered in the United States. One possible reason is that this pesticide has been related to large-scale bird mortalities in the past (H. Mendelssohn and U. Paz 1977, *Biol. Conserv.* 11:163–169).

In a follow-up study in 1996, we observed Swainson's hawks roosting in groves of exotic *Eucalyptus* sp. trees and feeding on grasshoppers (*Dichroplus* spp.) in sunflower and alfalfa fields in La Pampa Province. We surveyed approximately 2,500 km² and encountered large flocks of up to 12,000 hawks scattered throughout the area. In late January, we recorded four incidents of large-scale mortality with an estimated total of approximately 4,100 dead hawks. Chemical-use data from these incidents were obtained from landowners or applicators. Two incidents involved monocrotophos applications on alfalfa fields for grasshopper control. A total of 982 dead Swainson's hawks were found in fields and roosts adjacent to fields where the pesticide had been applied. In a third incident, 103 hawks were found dead after the OP dimethoate was sprayed on alfalfa for grasshopper control. In all three incidents, we found no age-class differences in mortality.

The largest incident of Swainson's hawk mortality occurred in a 120 ha alfalfa field sprayed with an unknown pesticide. An estimated 3,000 hawks were killed after this application. In this case, an estimated 75% of the dead hawks were adults. Overall results of this study indicate that continued large-scale mortalities from OP pesticide applications in Argentina wintering areas may threaten the future status of this species.

Based on past band recoveries and sightings (CIPA Seccion Argentina 1987, *Nuestras Aves* 13:13–16), as well as current data, wintering areas of the Swainson's hawk in Argentina include La Pampa, Buenos Aires, Cordoba, San Luis, and Sante Fe Provinces. The extent of the wintering range, however, has not been fully described. Crop production throughout much of La Pampa, Buenos Aires, and Cordoba consists largely of sunflower seed and alfalfa. Due to the area of overlap between these forms of agriculture and the high concentrations of Swainson's hawks in the area, it is likely that pesticide-related mortality may well exceed 5% of the world's population, 1% of which we recorded.

We are currently performing residue analyses on tissue samples of dead hawks. These data will lead to more conclusive evidence for the identity of the pesticides that led to the large-scale mortalities observed on the austral summer habitat of this species.

This investigation was supported by TIWET¹/ATRC,² USDA Forest Service, Boise State University, INTA,³ and the National Wildlife Federation. Special thanks go to M.J. Bechard, M.J. Hooper, T.E. Lacher, Jr., J.L. Panigatti, J.L. Garat, C. Peregalli and S. Salva.—**Michael I. Goldstein**, ¹**The Institute of Wildlife and Environmental Toxicology and** ²**Archbold Tropical Research Center, P.O. Box 709, Pendleton, SC 29670 U.S.A., Brian Woodbridge, USDA Forest Service, Klamath National Forest, 1312 Fairlane Dr., Yreka, CA 96097 U.S.A., Maria E. Zaccagnini and Sonia B. Canavelli, Wildlife Management Subprogram, ³Instituto Nacional de Tecnologia Agropecuaria, Estación Experimental Paraná, C.C. 128, Paraná, Entre Rios, Argentina, and Agustín Lanusse, C.C. #4, Estancia La Chanilao, 6207 Alta Italia, La Pampa, Argentina.**

BOOK REVIEWS

EDITED BY JEFFREY S. MARKS

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Books on Hawks and Owls: an Annotated Bibliography. By Richard R. Olendorff, Dean Amadon, and Saul Frank. 1995. Proceedings of the Western Foundation of Vertebrate Zoology, Vol. 6, No. 2. 89 pp., frontispiece. ISSN 0511-7550. Paper, \$10.00.—This compilation contains more than 600 citations of books and monographs from throughout the world that are devoted to falconiforms and strigiforms. Entries are listed alphabetically by author and cross-referenced for junior authors. The annotations range from long paragraphs to single sentences. Many of the longer annotations contain personal anecdotes that are available nowhere else. As a result, the bibliography is interesting to read in addition to being useful. An added bonus is the color frontispiece of a harpy eagle (*Harpia harpyja*) painted by Louis Agassiz Fuertes in 1899.

The treatment appears to be very complete for books (including several 1995 titles) but is less so for monographs. For example, three *Wildlife Monographs* are included (Clark 1975, McGarigal et al. 1991, Hayward et al. 1993), but four are excluded (Ellis 1979, Forsman et al. 1984, Swenson et al. 1986, McClelland et al. 1994). My only major criticism is that the bibliography contains no index. This is especially troublesome if one wishes to obtain a list of titles for a particular species. Granted, many of the books treat multiple species, and it would have been unwieldy to index each mention of a species. Nonetheless, an effort to index individual species and a few broad subject categories would have enhanced the utility of the bibliography. Despite this criticism, *Books on Hawks and Owls* is well worth obtaining. It is a fine testimony of the late Butch Olendorff's commitment to his profession.—**Jeff Marks, Montana Cooperative Wildlife Research Unit, University of Montana, Missoula, MT 59812 U.S.A.**

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The Wind Masters. By Pete Dunne. 1995. Houghton Mifflin Company, New York, NY. xvi + 263 pp., 66 scratchboard illustrations by David Sibley. ISBN 0-395-65235-9. Cloth, \$22.95.—This is a collection of vignettes (varying in length from 6–12 pages) on the 33 species of diurnal raptors that nest in North America north of Mexico. Included are the three cathartid vultures, although Dunne acknowledges that these birds are probably ciconiiforms rather than falconiforms. The Aplomado falcon (*Falco femoralis*) is omitted because the last documented nesting in the U.S. was in 1952. Each chapter is a fictional portrayal of an individual or pair of the species in question. The participants are placed in real situations where they perform plausible behaviors, but they “have a life beyond the disciplined standards that distinguish scientific treatments.” Thus, a male gyrfalcon (*Falco rusticolus*) waits hopefully for the return of its mate to a lonely cliff on Alaska's North Slope, a female Harris' hawk (*Parabuteo unicinctus*) bemoans the fact that her mate is an inept lover (although he's a “terrific” hunter), a black vulture (*Coragyps atratus*) hisses a few bars of *Teddy Bear's Picnic* while waiting for the morning thermals to develop, and a captive adult California condor (*Gymnogyps californianus*) dreams of the days when she soared as a free-flying juvenile.

Anthropomorphic accounts of wild animals seldom contribute anything of value and typically make me cringe. Such is not the case here. Dunne's portrayal of a gray hawk (*Buteo nitidus*) catching a lizard along Sonoita Creek brought me back to an afternoon more than a decade ago when I watched my first gray hawk, which had just caught a snake along that very same creek. His descriptions of the flight styles of short-tailed hawks (*Buteo brachyurus*) and white-tailed hawks (*B. albicaudatus*) left me with great disappointment that I have never seen these beautiful and distinctive species. Aside from the wonderful prose, each chapter contains

an informative tidbit or two about the natural history of the species depicted. For example, readers will learn of the white-tailed hawk's predilection for hunting along the edges of grass fires, the northern harrier's (*Circus cyaneus*) tendency for polygynous nesting, and the Swainson's hawk's (*Buteo swainsoni*) fondness for grasshoppers.

The book is not without fault, however. Dunne's description of mutualism between merlins (*Falco columbarius*) and their shorebird prey, and his assertion that territoriality benefits "populations" of harriers, exhibit an implicit acceptance of group selection theory. In several places he uses "juvenile" to describe "juvenal" plumages, and he does not consider tail feathers to be flight feathers (they are). The crested caracara should be placed in the genus *Caracara*, not *Polyborus*. Recent develop-

ments in DNA-DNA hybridization are attributed to *Fred* Sibley (rather than to Charles Sibley), and a red-tailed hawk's (*Buteo jamaicensis*) pituitary gland is allegedly stimulated into action by sunlight (the hypothalamus reacts to changes in photoperiod and stimulates the pituitary). At one point, "Ted" the fledgling peregrine falcon (*Falco peregrinus*) suddenly sports a "blue-gray back and helmeted head" of an adult. These criticisms are really minor quibbles. *The Wind Masters* is a wonderful book that is both informative and a joy to read. Written for a lay audience, it nonetheless will be cherished by open-minded professionals who appreciate good nature writing and are willing to accept that, perhaps condors *do* dream.—**Jeff Marks, Montana Cooperative Wildlife Research Unit, University of Montana, Missoula, MT 59812 U.S.A.**

THESIS ABSTRACT

ECOLOGY OF BALD EAGLES AT HUNGRY HORSE RESERVOIR, MONTANA

I documented bald eagle (*Haliaeetus leucocephalus*) nesting activity, behavior, habitat use, and human disturbance during 1985–88 at Hungry Horse Reservoir, northwestern Montana. All available records of bald eagle sightings (including migrating eagles) at the Reservoir were evaluated to help locate historic eagle-use sites and previous nesting territories. Only 13% of reported bald eagle sightings were adults during summer. Most records were of autumn migrants that foraged along the 100-km-long Reservoir or its inlet stream.

Two nest locations, apparently alternate sites on the same territory, were found. Productivity (young produced per occupancy) declined from 1.8 (1979–83) to 0.4 (1984–88). Durations of adult bald eagle visits to active nests in 1985 and 1986 averaged 42 min through the first week in July, but only 4 min thereafter despite differences in nestling age. The eagles nested in an old-growth stand and perched and roosted in large, old trees on an island or near the shoreline. Adults often flew to recently burned sites, where they soared on thermals rising from the blackened surface. Mountain whitefish (*Prosopium williamsoni*) and largescale sucker (*Catostomus macrocheilus*) were most frequently present in prey remains below perches.

Levels of lead, mercury, and cadmium in blood samples from 1985 and 1986 juveniles were within normal limits. Transmitters placed on the 1985 and 1986 juveniles from the Hungry Horse nest facilitated observation of post-fledging behavior and migration. After fledging, juveniles remained associated with the adults and the nest until early autumn, when they moved south across Montana. Both juveniles were near Dillon, Montana by 10 October. The 1985 juvenile was located near Cardston, Alberta, Canada, on 23 April 1986; it was with a group of migrating eagles traveling north.

Timber harvest and recreational activities precluded bald eagle use of several potentially important foraging areas. Eagles used areas well beyond previously established interim management zones. Information from this study provided a basis for preparation of a nest-site management plan for the U.S. Forest Service.—**Patricia T. McClelland. 1992. M.Sc. thesis, Wildlife Biology Program, University of Montana, Missoula, MT 59812 U.S.A.**

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The Journal of Raptor Research is distributed quarterly to all current members. Original manuscripts dealing with the biology and conservation of diurnal and nocturnal birds of prey are welcomed from throughout the world, but must be written in English. Submissions can be in the form of research articles, letters to the editor, thesis abstracts and book reviews. Contributors should submit a typewritten original and three copies to the Editor. All submissions must be typewritten and double-spaced on one side of 216 × 278 mm (8½ × 11 in.) or standard international, white, bond paper, with 25 mm (1 in.) margins. The cover page should contain a title, the author's full name(s) and address(es). Name and address should be centered on the cover page. If the current address is different, indicate this via a footnote. A short version of the title, not exceeding 35 characters, should be provided for a running head. An abstract of about 250 words should accompany all research articles on a separate page.

Tables, one to a page, should be double-spaced throughout and be assigned consecutive Arabic numerals. Collect all figure legends on a separate page. Each illustration should be centered on a single page and be no smaller than final size and no larger than twice final size. The name of the author(s) and figure number, assigned consecutively using Arabic numerals, should be pencilled on the back of each figure.

Names for birds should follow the A.O.U. Checklist of North American Birds (6th ed., 1983) or another authoritative source for other regions. Subspecific identification should be cited only when pertinent to the material presented. Metric units should be used for all measurements. Use the 24-hour clock (e.g., 0830 H and 2030 H) and "continental" dating (e.g., 1 January 1990).

Refer to a recent issue of the journal for details in format. Explicit instructions and publication policy are outlined in "Information for contributors," *J. Raptor Res.*, Vol. 27(4), and are available from the editor.

1996 ANNUAL MEETING

The Raptor Research Foundation, Inc. 1996 annual meeting will be held jointly with the American Ornithologists' Union annual meeting on 13–17 August at Boise State University, Boise, Idaho. Details about the meeting and a call for papers will be mailed to Foundation members in the spring of 1996 and can be obtained from Peter Lowther, Scientific Program Chairman, Field Museum of Natural History, Roosevelt Road at Lakeshore Drive, Chicago, IL 60605-2496, (telephone 312 922-9410 ext. 461; Fax 312 922-2572; e-mail lowther@fmnh.org) and Marc Bechard and Alfred Dufty, Local Co-chairs, Department of Biology, Boise State University, Boise, ID 83725 (telephone 208 385-3262; Fax 208 385-3006; e-mail rbibecha@idbsu.idbsu.edu or adufty@claven.idbsu.edu).

RAPTOR RESEARCH FOUNDATION, INC., AWARDS

Recognition for Significant Contributions¹

- The **Dean Amadon Award** recognizes an individual who has made significant contributions in the field of systematics or distribution of raptors. Contact: **Dr. Clayton White, 161 WIDB, Department of Zoology, Brigham Young University, Provo, UT 84602 U.S.A.** Deadline August 15.
- The **Tom Cade Award** recognizes an individual who has made significant advances in the area of captive propagation and reintroduction of raptors. Contact: **Dr. Brian Walton, Predatory Bird Research Group, Lower Quarry, University of California, Santa Cruz, CA 95064 U.S.A.** Deadline: August 15.
- The **Fran and Frederick Hamerstrom Award** recognizes an individual who has contributed significantly to the understanding of raptor ecology and natural history. Contact: **Dr. David E. Andersen, Department of Fisheries and Wildlife, 200 Hodson Hall, 1980 Folwell Avenue, University of Minnesota, St. Paul, MN 55108 U.S.A.** Deadline: August 15.

Recognition and Travel Assistance

- The **James R. Koplín Travel Award** is given to a student who is the senior author of the paper to be presented at the meeting for which travel funds are requested. Contact: **Dr. Petra Wood, West Virginia Cooperative Fish and Wildlife Research Unit, P.O. Box 6125, Percival Hall, Room 333, Morgantown, WV 26506-6125 U.S.A.** Deadline: established for conference paper abstracts.
- The **William C. Andersen Memorial Award** is given to the student who presents the best paper at the annual Raptor Research Foundation Meeting. Contact: **Ms. Laurie Goodrich, Hawk Mountain Sanctuary, Rural Route 2, Box 191, Kempton, PA 19529-9449 U.S.A.** Deadline: Deadline established for meeting paper abstracts.

Grants²

- The **Stephen R. Tully Memorial Grant** for \$500 is given to support research, management and conservation of raptors, especially to students and amateurs with limited access to alternative funding. Contact: **Alan Jenkins, George Miksch Sutton Avian Research Center, Inc., P.O. Box 2007, Bartlesville, OK 74005-2007 U.S.A.** Deadline: September 10.
- The **Leslie Brown Memorial Grant** for \$500–\$1,000 is given to support research and/or the dissemination of information on raptors, especially to individuals carrying out work in Africa. Contact: **Dr. Jeffrey L. Lincer, Sweetwater Environmental Biologists, Inc., 3838 Camino del Rio North, Suite 270, San Diego, CA 92108 U.S.A.** Deadline: September 15.

¹ Nominations should include: (1) the name, title and address of both nominee and nominator, (2) the names of three persons qualified to evaluate the nominee's scientific contribution, (3) a brief (one page) summary of the scientific contribution of the nominee.

² Send 5 copies of a proposal (≤5 pages) describing the applicant's background, study goals and methods, anticipated budget, and other funding.